Eastern Deciduous Forest Restoration Goals and Methods: A Literature Synthesis

Prepared for
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TABLE OF CONTENTS

1. INTRODUCTION

The purpose of this document is to review and synthesize literature to inform ecological restoration being conducted by The Nature Conservancy and project partners in the Southern Lake Champlain Valley region. Although the primary focus of the review is knowledge about eastern deciduous forest restoration, with particular emphasis on the clayplain and floodplain forests of eastern New York and Vermont, relevant literature from other ecosystem types and regions is included. The literature review includes four main categories of information: 1) establishing goals for and evaluating the success of restorations; 2) restoration strategies and techniques; 3) ecosystem structure and processes and forest restoration; and 4) the composition, structure, and dynamics of Champlain Valley clayplain and floodplain forests.

2. METHODS

A literature search and synthesis was conducted for this project between February and May 2003. A variety of books, journals, bibliographic archives and databases were searched to obtain the best available and most up-to-date peer-reviewed and gray literature on restoration. A thorough search was conducted of at least the last ten years of the major restoration periodicals published in the U.S.: Restoration Ecology, and Ecological Restoration and its predecessor Restoration and Management Notes. Bibliographic indices and databases searched include BIOSIS (comprehensive databases of life science information), Agricola (The citation database of the National Agriculture Library), JSTOR (full text journal archive), and USDA Forest Service publication archives (www.fs.fed.us/ne/newtown_square/publications; sirsi.fs.fed.us; www.srs.fs.usda.gov). Other journals that were searched selectively, but are not indexed in the above databases include Applied Vegetation Science and The Natural Areas Journal. In addition, the catalogs of local college and university libraries were searched for relevant volumes. Online publications were procured from websites including the Society for Ecological Restoration (www.ser.org), Federal Interagency Stream Corridor Restoration Working Group (www.usda.gov/stream_restoration), and NRCS Conservation Practice Standards (www.ftw.nrcs.usda.gov). Telephone interviews were conducted with individuals involved in The Big Woods Project in Minnesota. Ecologists from the Vermont, New York and New Hampshire Natural Heritage Programs and the Boston office of NatureServe were contacted to obtain available data and summary information on floodplain forest community types in the region.

A bibliographic database was created for the project and the final version uses Endnote™ (ISI Research) and contains approximately 435 citations. For each reference, there is a complete citation. For many references an abstract, notes about how the reference was found, and a URL (for full text or abstract) are included. A standard list of keywords was developed (See Appendix A) and for most references, keywords were entered into the keywords field. The database was provided to TNC along with this report; new citations, keywords, and additional information can be added.

3. ESTABLISHING GOALS AND EVALUATING THE SUCCESS OF RESTORATION

This section discusses some ideas and concepts pertinent to setting goals, planning, and evaluating ecological restoration projects. There is widespread agreement that more effective goal-setting, planning and assessment of restoration projects is needed (Berger 1991, Aronson et al. 1995, Hobbs and Norton 1996, Michener 1997, FISRWG 1998, Kondolf 1998; Allison 2002, Hargrove et al. 2002). Commonly, restoration projects fail to identify realistic and achievable goals, and an adequate understanding of the processes causing degradation has not been developed (Hobbs and Norton 1996, Mitsch and Gosselink 2000, Allison 2002). Frequently, there are inadequate resources or means for evaluation (Hobbs and Norton 1996, Aronson et al. 1995, Michener 1997, FISRWG 1998, Kondolf 1998). Furthermore, it has been argued frequently that the conceptual focus of restoration is too narrow and restoration is not incorporated into broad land-use planning (Hobbs and Norton 1996). Restoration planning is explored further in the following subsections: 1) Definitions and concepts; 2) Current paradigms of succession and dynamics; 3) Habitat degradation and restoration thresholds; 4) Reference conditions; 5) Defining reasonable goals; 6) Measuring attributes and setting performance standards; 7) Spatial considerations; 8) Hypothesis testing and experimentation: 9) Monitoring; 10) Adaptive management; 11) Measuring success; 12) Temporal considerations; 13) Social, cultural and economic considerations; and 14) Recommendations and guidelines.

Definitions and Concepts

There has been an increasing awareness of the need for a broad and realistic definition of ecological restoration that allows for a range of goals, applied in a variety of different circumstances and over broad landscapes. Thus, contemporary thinking (Aronson et al. 1995, Hobbs and Norton 1996, Apfelbaum and Chapman 1997, White and Walker 1997, Stanturf et al. 1998, Hobbs and Harris 2001, SER 2002) has moved away from a strict definition of ecological restoration as recreating self-maintaining ecosystems of a specific predisturbance type (see Cairns 1989, NRC 1992, and SER 1993 cited in Covington et al. 1999). Recently the Society of Ecological Restoration (SER 2002) defined ecological restoration as “the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed.” This definition is not unlike the ideas put forth by Hobbs and Norton (1996) who stated that the aim of ecological restoration is to return a degraded system to a productive or aesthetically pleasing or valuable system (from a conservation perspective) and one that is sustainable in the long term. Similarly, Apfelbaum and Chapman (1997) also emphasized the importance of sustainability and described ecological restoration as a “practical management strategy that uses ecological processes in order to maintain ecosystem composition, structure and function with minimal human intervention.”

A narrow definition of ecological restoration that focuses strictly on restoring self-maintaining, predisturbance structure and function is often seen as limiting (Bratton 1992, Hobbs and Norton 1996, Apfelbaum and Chapman 1997, Ehrenfeld 2000). Hobbs and Harris (2001) suggested that “if we change the focus of restoration from trying to recreate something from the past to trying to repair damage and creating systems which fulfill sensible goals, we will go a long way to solving many of the conundrums facing the science and practice of restoration ecology." It is often difficult to define historical conditions, and species associations are variable over time (Hobbs and Norton 1996, Ehrenfeld 2000, Allison 2002, SER 2002). Restoring all aspects of the former composition and structure is neither practical nor economically feasible (Apfelbaum and Chapman 1997). New assemblages of species may arise that are valuable for biodiversity and productivity (Bratton 1992, Hobbs and Norton 1996). Moreover, since key species or processes may be missing or not able to be sustained on their own, managers must often intervene to maintain a system, making the goal that restored areas be self-maintaining unrealistic (Bratton 1992, Pickett and Parker 1994, Covington et al. 1999).

It has been suggested that restoration activities should be restricted to relatively undisturbed ecosystems and to those with the greatest opportunity for success (Cairns 1989, NRC 1992). A number of authors countered that restoration should encompass a variety of projects along a continuum from restoring the productivity of totally devastated sites to the management of relatively modified sites (Westman 1991, Hobbs and Norton 1996). Hobbs and Norton (1996) argued if restoration is to take a landscape perspective, the most degraded ecosystems need to be considered in restoration projects, and Hargrove et al. (2002) stated that restoration is only an efficient conservation tool in more highly fragmented landscapes.

Rather than focusing on restoring areas to a static endpoint, contemporary thinking focuses on the importance of restoring ecological processes. Some authors maintained that restoration should work to restore ecological processes such that an ecosystem is either returned to its developmental trajectory toward a chosen state (Covington et al. 1999, SER 2002) or moved away from a human-induced trajectory (White and Walker 1997). Depending on the aim of the restoration, various states could be identified, such as presettlement conditions, pre-human conditions, or a traditional agricultural ecosystem (SER 2002). Hobbs and Norton (1996) stressed the need to restore ecological processes and asserted that a goal of reaching some historic natural conditions may not be attainable and greater emphasis should be placed on dynamic goals that take into account the "changing nature of the environment." Along the same lines, Apfelbaum and Chapman (1997) maintained, "Ecological restoration seeks to reestablish an ecosystem's capacity to support species diversity, internal processes and resiliency in the face of changing conditions." The primary goal of woodland creation in Great Britain is to allow a species-rich, more-or-less-natural woods to develop over many decades (Spencer 1995), and Spencer posed the tough question of how to succeed in that in the face of continuing changes in society.

Current Paradigm of Succession and Dynamics

It is important to consider current ecological thinking about succession when planning restoration projects, as speeding up or directing succession is a prevailing focus of restoration (Hobbs and Norton 1996). Of particular relevance to restoration is the recognition that, due to differences in site history, human influences, context and dynamic processes, multiple pathways of succession are possible leading to alternative stable states (Westman 1991, Pickett and Parker 1994, Hobbs and Norton 1996, SER 2002). Thus, the outcomes of a particular restoration approach may vary considerably when carried out at different locations although initial conditions may appear similar (Hobbs and Norton 1996). Conceptual models of succession are particularly relevant to restoration in assessing the value of reference conditions and defining achievable goals for the restoration (Pickett and Parker 1994, Aronson et al. 1995, Hobbs and Norton 1996). A review of the vast literature regarding succession was beyond the scope of this paper.

Habitat Degradation and Restoration Thresholds

The idea of "restoration thresholds" is a useful concept in helping to understand how to assist in the recovery of degraded ecosystems (Hobbs and Norton 1996, Hobbs and Harris 2001, and others). The idea is based on the state and transition modeling approach that depicts the potential for ecosystems to exist at a variety of alternative stable states (Yates and Hobbs 1997).

Degrading influences – "stressors" – can force a transition to a degraded state. Due to restoration thresholds, caused by biotic interactions or abiotic limitations (Whisenant 1999, cited in Hobbs and Harris 2001), the system may be prevented from recovering from the degraded state without outside manipulation (Hobbs and Norton 1996, Hobbs and Harris 2001, SER 2002). Soil erosion, competition from invasive species, an altered disturbance regime, and heavy grazing are examples of stressors that can cause a system to cross a threshold. If the transition to a degraded state is cause by biotic interactions (such as grazing pressure), then biotic manipulations such as removing the grazing animals and planting vegetation may be sufficient to restore the system. If there is a physical limitation, however, such as soil degradation in addition to grazing pressure, the physical alteration must also be addressed if manipulating the biota is to be effective. It is important to identify the <u>fundamental</u> cause of degradation, define restoration goals based on that understanding, and select actions that will restore ecosystem functioning (Hobbs and Harris 2001). As an example, it is frequently emphasized in wetland restoration that the appropriate hydrology must be in place before the native vegetation can recover (NRC 1992, Mitsch and Gosselink 2000).

According to Covington et al. (1999), an important concept in guiding the restoration of degraded ecosystems is the "evolutionary environment." The idea is based on the understanding that species adapt to, and come to depend on, a set of conditions over the course of their evolution. It is assumed that when a factor has been of sufficient intensity and duration to exert selection pressure such that the species is adapted to it, the factor becomes a part of the species evolutionary environment. Human activities such as soil degradation, transport of exotic species, and change in disturbance regime can be seen as disrupting evolutionary trajectories and causing degradation of ecosystems. Thus, defining the historic "range of natural variation" that characterized the species evolutionary environment is important so that ecological restoration can reverse ecosystem degradation and set ecosystems on a trajectory more in keeping with their evolutionary environment.

Reference Conditions

Reference information is often used in restoration planning for 1) establishing site-specific restoration goals (Aronson et al. 1995, Mitsch and Gosselink 2000, Clewell et al. 2000, Stanturf et al. 2001, SER 2002), 2) determining the restoration potential of sites (Apfelbaum and Chapman 1997, White and Walker 1997), and 3) evaluating the success of a restoration (White and Walker 1997, Mitsch and Gosselink 2000, Clewell et al. 2000, SER 2002) (See also Appendix B). Some authors have warned that while a model of some sort is necessary for guiding all restoration projects, reference information can have limited usefulness for evaluating project success (Hobbs and Norton 1996, Stanturf et al. 2001), and, perhaps more significantly, that reference information can potentially result in establishing unattainable goals (Hobbs and Norton 1996). Pickett and Parker (1994) cautioned against the assumption that there is "one reference stand or system that can inform restoration." Given this debate, it seems wise to understand the potential value and limitations of reference information so that reference information can be used judiciously and effectively (White and Walker 1997).

The following are some criticisms or limitations of various types of reference information. Reference information often fails to encompass all the variability in potential trajectories and natural range of variability in ecosystem attributes (Pickett and Parker 1994, Hobbs and Norton 1996, Stanturf et al. 1998). Frequently, it may be difficult or impossible to determine with accuracy all the potential trajectories especially given the information that can be obtained considering the resources, time, and funding that are usually available (SER 2002). Reference stands represent a snap-shot view in time (White and Walker 1997); thus, if a reference ecosystem represents a later developmental stage, it may be necessary to interpolate back to an earlier developmental stage for project planning and evaluation (SER 2002). Identifiable reference stands will likely have suffered some adverse human impacts (including unknown or unmeasurable rare events) and thus do not represent conditions that should be emulated in the restored site (Stanturf et al. 1998, SER 2002). Problems with historical data are that unmeasured factors may make historical changes difficult to interpret, and historical information has poor spatial and temporal resolution (White and Walker 1997). Information such as practical experience and experimentation achieved through wetland creation, rotational grazing, and mine reclamation may be of greater value for informing restoration (Apfelbaum and Chapman 1997). The value of the reference site may be dependent on the distance from the restored site due to variability in conditions (White and Walker 1997). Finally, established techniques available for integrating the various types of reference information are limited (White and Walker 1997).

The shortcoming of using one reference stand to guide a restoration is well established. It is thus recommended that a description of the "reference ecosystem" be a composite of attributes from several sites and a variety of different types of information including baseline information from the site to be restored (White and Walker 1997, Clewell et al. 2000, SER 2002). It is recommended that the reference information contain detail on both biotic and abiotic features and seral stages (Clewell et al. 2000). Kondolf (1998) suggested that for each restoration project there should be a geomorphically suitable goal or end state that serves as a "leitbild" or guiding image. The "leitbild" can be a composite of observations from multiple sites with similar geomorphic characteristics and the image can be modified based on human alteration to the site to be restored.

White and Walker (1997) summarized four types of reference information. The following describes the kind of information that could be assembled for each type of reference data to create a composite description of important attributes. For more detail of the limitations of each type of data, see White and Walker (1997).

1. <u>Contemporary on-site information</u> – direct evidence of successional trends, change in processes (relict fire-dependent species, absence of reproduction), change in components (exotic species, absent species), process legacies (i.e., meander bends in rivers), current conditions (soil, hydrology, species composition, environmental conditions).
2. <u>Contemporary reference site data</u> – direct evidence of how measured processes influence composition, structure, and dynamics under current environmental conditions, a means of testing hypotheses and conducting controlled experiments.

3. Historical on-site data – direct evidence of variables that require restoration (gleaned from aerial and ground photos, oral and written histories, herbarium and museum specimen records, paleoecological evidence (e.g., fossil pollen, charcoal)).
4. Historical reference site data: view of past states, variability of human influences.

Setting Goals

The process of establishing goals is essential to the success of restoration and contributes to guiding the development and implementation of the project, as well as measuring the progress and success of the project (Hobbs and Norton 1996, FISRWG 1998, Kenna et al. 1999, Hobbs and Harris 2001, Hargrove et al. 2002, SER 2002). Goals should be site-specific and defined based on an understanding of the scope and limitations of the project. Thus, different sets of goals are appropriate to reflect a diversity of project aims and spatial scales (Hobbs and Norton 1996, FISRWG 1998, Kenna 1999, Ehrenfeld 2000, Hargrove et al. 2002). Project limitations include technical limitations (including availability of data), economic constraints, social factors (land-use conflicts, permits) and physical constraints (FIWRWG 1998, Kenna et al. 1999). Goals should be kept as simple as possible to better enable their measurement (FISRWG 1998, Mitsch and Gosselink 2000, Hobbs and Harris 2001).

Goals should stem from a rigorous understanding of the current state of an ecosystem or landscape and the underlying factors that contribute to current conditions (Hobbs and Norton 1996, Apfelbaum and Chapman 1997, FISRWG 1998, Covington et al. 1999, Mitsch and Gosselink 2000, Hobbs and Harris 2001). Such an understanding helps determine what restoration activities are possible and leads to a definition of the restoration goals (Hobbs and Harris 2001). Apfelbaum and Chapman (1997) emphasized that precise goals should be established after collecting information and formulating hypotheses about the factors that caused the current conditions and their significance to the development of future states. Quantitative or qualitative goals established for each management unit should reflect the best information on restoration potential and also specify the desired future condition(s). Goals should be altered when new information becomes available (Apfelbaum and Chapman 1997). As an illustration, initial data collected on altered wetland processes (specifically hydrology, sedimentation (increased deposition), nutrient cycling (increased N and P inputs), and structure (monocultures of cattails and reed canary-grass)) might lead to testable hypotheses about the relationship between the degraded conditions and loss of native species. Actions might be aimed at specific efforts to restore the hydrological dynamics and reduce nutrient runoff. Therefore, specific goals could be formulated with an understanding of the structure and function of reference wetlands and site conditions that serve as models for desired future condition.

There is considerable debate regarding the extent to which goals should specify desired future conditions (see "Definitions and conceptual ideas" and "Reference conditions" above). Hobbs and Norton (1996) stressed that for the same location, multiple potential conditions are possible, and thus restoration goals that are dynamic are required for restorations to be successful. Aronson et al. (1995) countered that to develop and evaluate a project, it is necessary to determine “at the outset some standard for comparison” (such as the “leitbild” or guiding image developed from multiple sites as suggested by Kondolf (1998)). Aronson et al. (1995) maintained that a standard for comparison is necessary to “clarify research goals and methodology,” but that this standard need not be rigid, as the goals and concept of the reference most likely will be modified before the end of the project.

Although the specificity of the desired future condition varies among projects. Rather than achieving the desired condition in the short term, a feasible goal may be to work to get a site into a condition from which it will then be able to develop toward a desired condition over the long term. In any case, Hobbs and Norton (1996) recommended that a first step in establishing goals for restoration is inferring the broad vegetation type that would have occurred at the site from soils, landform, and other biophysical features.

Goals that address a combination of structural and functional features are often recommended for ecological restoration (Berger 1991, NRC 1992, Mitsch and Gosselink 2000). Many authors have maintained that goals recognizing social values are a critical component of restoration planning (Berger 1991, Westman 1991, Apfelbaum and Chapman 1997, Covington et al. 1999, Hobbs and Harris 2001, SER 2002). In some cases, an appropriate goal may be to try to restore structure or composition. In other situations, the restoration of productivity in a highly degraded area may be a more reasonable goal and composition will not be as important (Hobbs and Harris 2001). Hargrove et al. (2002) argued that it is important that ecosystem processes do not become the end goal at the expense of species diversity. It is interesting to note that much restoration work in North America has been aimed largely at the goal of restoring “presettlement” ecosystems, while a more common focus in other parts of the world with a longer history of settlement (such as Europe) has been to restore cultural ecosystems (SER 2002) such as grasslands (Rosen and van der Maarel 2000).

In wetland creation and stream restoration, the goals often focus on providing ecosystem services including: bank stabilization, flood control, wastewater treatment, stormwater or nonpoint-source pollution control, water quality, wildlife enhancement, fisheries enhancement, creating habitat for aquatic and terrestrial species, upland soil and water conservation, creating research wetlands, and aesthetics (FISRWG 1998, Mitsch and Gosselink 2000). In thinking about goals, it seems useful to consider three key lessons learned from woodland creation in Great Britain: 1) woodland creation is fated to success, 2) attempts to recreate ancient woodland are doomed to failure, and 3) outcomes are unpredictable (Spencer 1995). Additionally, planners should consider that “fashions” change over time, and that future personnel, technology, information, and funding will be different that those currently available (Spencer 1995), hence the need for adaptive management that includes adaptive goal-setting.

Measuring Attributes and Establishing Performance Standards

The specific attributes to be measured should be determined by the types of goals and actions required to achieve restoration. Ehrenfeld (2000) categorized three major types of restoration projects that have resulted in developing different sets of goals and measuring different attributes: 1) restoration of species composition, 2) restoration of ecosystem services, and 3) restoration of whole landscapes.

Hobbs and Norton (1996) defined six categories of ecosystem attributes that characterize natural ecosystems and can help to identify goals for a restoration. These include: 1) composition - species presence and relative abundance, 2) structure - vertical arrangement of vegetation and soil, 3) pattern - the horizontal arrangement of an ecosystem, 4) heterogeneity - the variation in composition, structure, and pattern, 5) function - the performance of basic ecological processes, and 6) dynamics and resilience - successional processes and ability to recover from disturbance.

The Society of Ecological Restoration has put forth the following is list of potential attributes of restored ecosystems (SER 2002):

1. The restored ecosystem contains a characteristic assemblage of the species that occur in the reference ecosystem and that provide appropriate community structure.
2. The restored ecosystem consists of indigenous species to the greatest practicable extent. In restored cultural ecosystems, allowances can be made for exotic domesticated species and for non-invasive ruderal and segetal *(growing intermixed with crops*) species that presumably co-evolved with them.
3. All functional groups necessary for the continued development and/or stability of the restored ecosystem are represented or, if they are not, the missing groups have the potential to colonize by natural means.
4. The physical environment of the restored ecosystem is capable of sustaining reproducing populations of the species necessary for its continued stability or development along the desired trajectory.
5. The restored ecosystem apparently functions normally for its ecological stage of development, and signs of dysfunction are absent.
6. The restored ecosystem is suitably integrated into a larger ecological matrix or landscape, with which it interacts through abiotic and biotic flows and exchanges.
7. Potential threats to the health (*based on a set of ecosystem attributes*) and integrity (*based on a set of biodiversity characteristics*) of the restored ecosystem from the surrounding landscape have been eliminated or reduced as much as possible.
8. The restored ecosystem is sufficiently resilient to endure the normal periodic stress events in the local environment that serve to maintain the integrity of the ecosystem.
9. The restored ecosystem is self-sustaining to the same degree as its reference ecosystem, and it has the potential to persist indefinitely under existing environmental conditions. Nevertheless, aspects of its biodiversity, structure, and functioning may change as part of normal ecosystem development and may fluctuate in response to normal periodic stress and occasional disturbance events of greater consequence. As in any intact ecosystem, the species composition and other attributes of a restored ecosystem may change as environmental conditions change.

Other attributes may be added to the list based on the goals of the restoration (See Appendix C). For example, goals may include habitat for rare species, aesthetic qualities, means for public involvement in restoration, or supplying goods and services in a sustainable way for society (SER 2002).

It is recommended that performance criteria for measuring each selected attribute be determined at the outset. Performance criteria are linked to the actual goals and parameters that are to be measured, and they establish the limits of the parameters (FISRWG 1998, SER 2002). The project budget and resources dictate the degree to which attributes are measured; some parameters may be assessed directly, while others, including many ecosystem functions, can often only be evaluated indirectly (SER 2002). Research is needed to identify attributes that can be readily measured to serve as indicators of processes. For example, litter-fall might be an indicator of above-ground productivity (Westman 1991).

According to Westman (1991), more than one parameter should be selected to monitor each major biotic and physical feature; for example, biomass and productivity might both be included. Ideally, parameters should be selected to have low levels of intercorrelation. A contrasting recommendation is that a minimal number of parameters be selected, making sure that parameters measured have established performance criteria so that the goals of the restoration can be evaluated (NRC 1992, FISRWG 1998, SER 2002).

Spatial Considerations

The benefits of conducting ecological restoration at the landscape scale while simultaneously considering small-scale processes are widely touted (Hobbs and Norton 1996, Apfelbaum and Chapman 1997, FISRWG 1998, Wissmar and Beschta 1998, Kenna et al. 1999, Ehrenfeld 2000, Hargrove et al. 2002). (Landscape-scale work considers the flows, interactions, and exchanges between adjacent ecosystems (Forman 1995)). Nevertheless, due to the complexity of ecosystems, the process of making recommendations to guide restoration at the landscape scale is in its infancy (Hobbs and Norton 1996, Ehrenfeld 2000). General guidelines acknowledge the benefits of buffers, corridors, and large patches, and advise “bigger is better than smaller,” and “connected is better than isolated” (Forman 1995). Site-specific objectives can provide insight into what landscape-level priorities might be (Hobbs and Norton 1996). For example, if an adjacent agricultural field will be abandoned, it may not be a priority to create a wetland for the goal of controlling non-point pollution (Mitsch and Gosselink 2000). Moreover, asking whether restoration thresholds (such as altered hydrology or severe fragmentation/lack of connectivity) exist at the landscape scale could be a fruitful approach for identifying priorities on the landscape (Hobbs and Harris 2001).

Hargrove et al. (2002) argued that even though there are many disadvantages to undertaking restoration (including expense, time and uncertainty), restoration of degraded habitat might be the only way to meet large-scale (e.g., ecoregional) conservation goals in highly fragmented or degraded landscapes. The most efficient use of conservation dollars, according to Hargrove et al. (2002), is to protect intact remnants. Restoration should not be undertaken to create new exemplary natural communities (“element occurrences”), rather restoration should focus on supporting existing remnants and improving landscape context and landscape connectivity. Specifically, restoration should improve at least one of the following components of the conservation target: size, condition, or landscape context (Hargrove et al. 2002).

Hypothesis Testing and Experimentation

Because we often lack a clear understanding of complex ecological interactions, experimentation that can guide restoration is recommended as a key component of the process (Apfelbaum and Chapman 1997, FISRWG 1998, Covington et al. 1999, Kenna et al. 1990, Mitsch and Gosselink 2000, Hargrove et al. 2002). Goals can be restated as a set of testable hypotheses, and performance criteria can be used to evaluate aspects of the restored system in order to assess progress toward goals (FISRWG 1998). In addition, science can benefit from testing ideas about ecological phenomena, and it is critical that results are shared so others can benefit from the knowledge gained (Pickett and Parker 1994, Michener 1997). Clewell and

Rieger (1997) identified types of information and experimentation that would benefit restoration practitioners (See Appendix D).

Apfelbaum and Chapman (1997) outlined a procedure for developing testable hypotheses. Hypotheses originating from the initial information collected should be developed regarding composition, structure, and function; they should then be revised in light of the technical literature and reconnaissance visits to remnant ecosystems. Hypotheses should offer an explanation of how stressors led to the current conditions and the influence these stressors might have on future states. Applying similar management treatments to restoration units with similar conditions can test hypotheses.

Covington et al. (1999) provided a good example of how experiments can be designed to test hypotheses about restoration needs. They put forth an example from southwestern ponderosa pine (*Pinus ponderosa*) ecosystems where grazing, post-settlement fire suppression, and logging have resulted in altered structure and function. Symptoms of the degraded conditions include: increased fuel accumulation, decreased diversity of herbs and woody species, decreased soil moisture and nutrient availability, increased mortality of older trees, decreased stream flows, and increased fire severity and size. Covington et al. (1997, cited in Covington et al. 1999) described ecological restoration experiments designed to test the following hypotheses based on observed conditions:

1) Both restoration of ecosystem structure and reintroduction of fire are necessary for restoring rates of decomposition, nutrient cycling, and net primary productivity to natural (presettlement) levels; and
2) The rates of these processes will be higher in an ecosystem that is operating within some facsimile of its natural structure and disturbance regime.

Hypotheses were tested in replicated small plot studies and parameters to be measured were based on specific hypotheses. The research required a long-term commitment, 24 years, in which eight cycles of three-year burn intervals were conducted.

Generating multiple testable hypotheses can be an effective approach for restoration (Michener 1997). In a statement from 1897, Chamberlin (1995, cited in Michener 1997) described the process of generating multiple working hypotheses:

> "The effort is to bring into view every rational explanation about the phenomenon in hand and to develop every tenable hypothesis relative to its nature, cause, or origin, and to give all of these as impartially as possible a working form and a due place in the investigation."

Walters and Holling (1990) described a rendition of the multiple working hypothesis idea that is relevant to adaptive/experimental management. Their "active-adaptive" approach uses available data to develop a variety of alternative models and policy choices (or restoration actions). Multiple alternate policy choices (or actions) are selected from the models that were developed. The selections strike a balance between expected short-term performance and the benefit of knowing which alternative model is best. They contrast this approach with the "passive-adaptive" experimental approach, in which a single best estimate model is used to develop the policy (or restoration actions). In this case, there is no way of knowing if the selected model is the best. Their third approach is the "trial and error approach," in which initial choices are haphazard and later decisions are selected from a subset that yields the best results. Walters and Holling appear to favor the "active-adaptive" approach because: 1) formulating alternative models requires fruitful exchange between practitioners and scientists, 2) critical evaluation of different models requires careful scrutiny of available information, and 3) imaginative actions may be developed in order to avoid less-focused and risky experimentation. Moreover, the passive-adaptive approach cannot distinguish between management and environmental effects, as all possible models are not tested. At the outset, relatively small portions of the landscape can be devoted to test techniques where the response is most uncertain (Kenna et al. 1999).

There are a number of difficulties and challenges associated with developing experiments as a component of restoration. Among these are: 1) long-term study is often required (Michener 1997, Covington et al. 1999), 2) studies of interest to scientists may have limited value to the restoration (Michener 1997), 3) scientists may hesitate to prescribe restoration actions without rigorous scientific testing with experimental controls and replication, 4) the theoretically optimum, controlled, experimental design with replication can rarely be met when designing experiments as part of restoration (Michener 1997), and 5) tradeoffs must be made between the number of parameters sampled versus the spatial and temporal resolution (Michener 1997).

Michener (1997) offered information concerning how some of the challenges of designing experiments as a part of restoration projects could be overcome and also offered advice on statistical approaches for dealing with the types of data that are generated (see Michener 1997 for more detail). In short, he asserted that experiments to guide restoration can be informed by designs that have been developed to experimentally evaluate events like floods, oil spills, and fires, which are large, unplanned, unreplicated, and lack controls. Research approaches that can be useful include: space-for-time substitution as an alternative to long term sampling (Pickett 1989), simulation modeling, long-term studies, large-scale comparative studies, and focused experimentation to identify causal mechanisms (Michener 1997).

Statistical techniques that have value to restoration include a variety of simple descriptive statistics as well as diversity indices, cluster analysis, and various ordination techniques appropriate for characterizing patterns before and after restoration. An example of an analytical design developed to evaluate conditions before and after an oil spill is a technique called BACI (before – after- control –impact), which uses data from a single treatment site and a single control site collected before and after treatment. It is also noted that geostatistics can be used to address the problems with spatially autocorrelated data.

In designing experiments, Michener (1997) advises that is particularly important to pay careful attention to presuppositions, such as site history, and to place the restoration project within a broader context. As in all long-term studies with multiple investigators, careful attention must be paid to data management issues (Apfelbaum and Chapman 1997, Allison 2002), including careful documentation (meta-data) of data content, context, quality, and structure (Michener 1997).

Monitoring

Monitoring is an important component of the restoration process, and it is worth noting that the monitoring phase is defined differently by different authors (see Appendix B). Apfelbaum and Chapman (1997) stated that the monitoring component of restoration should be designed after the goals have been established so that the effectiveness of the restoration strategy can be effectively measured with carefully selected parameters that reflect the specific goals. They distinguish between the “gathering information phase,” where on-site, historical, and other reference information is collected before goals are established, and the “monitoring phase,” which is initiated after goals are developed. They note that all aspects of the restoration should be planned before monitoring to ensure that the parameters that are chosen can effectively measure progress toward goals. It has also been recommended that the timing, frequency, intensity, and duration of sampling be established, and data analysis methods determined, during the project planning phase (NRC 1992, FISRWG 1998, SER 2002). In contrast, other authors did not distinguish between a “gathering information phase” and “baseline monitoring.” According to the National Research Council (1992) and the Federal Interagency Stream Restoration Working Group (1998), baseline monitoring provides information essential for both establishing goals and objectives and providing a basis for evaluating the completed restoration. While there may be some overlap between the gathering information and monitoring phases (depending on how much is known about conditions and expected goals before the start of the project), it makes sense that these two phases be distinguished as they have separate purposes.

Van Horn and Van Horn (1996) presented a simple quantitative and qualitative method for monitoring vegetation structure and composition based on photopoints. Photographs taken with a “density board” (something like a broad stadia rod, with alternating color bands) in the background were used to rapidly vegetation structure and composition (in relatively coarse terms) at different time periods. The photo is a qualitative representation of the vegetation composition and structure, while visual estimates of vegetation parameters recorded per “band” on the density board provided quantitative metrics.

Adaptive Management

Adaptive management in restoration consists of using the results of monitoring and experimentation during the restoration process to alter goals and re-design restoration treatments (FISRWG 1998, Covington et al. 1999). It is almost a certainty that new information learned during the course of a restoration will challenge previously held hypotheses and restoration goals (Apfelbaum and Chapman 1997). Adaptive management is a component of the Northwest forest plan, where due to the complexity of predicting the outcome of management actions, new information from research and evaluation are used to adjust objectives and improve the implementation and achievement of goals (Covington et al. 1999). It seems wise to acknowledge and communicate early on the uncertainty of restoration (Cairns 1989), to view restoration as a dynamic process (Pickett and Parker 1994, Hobbs and Norton 1996), and to embrace the value of the adaptive management model (Apfelbaum and Chapman 1997, FISRWG 1998, Kenna et al. 1999, Hargrove et al. 2002).

Measuring Success

Acknowledging the dynamic nature of ecosystems and the experimental, dynamic, and adaptive nature of restoration suggests that restoration success should not be hinged on whether initial goals are achieved, but rather on the effectiveness of the restoration process itself (Pickett and Parker 1994, Hobbs and Norton 1996, Apfelbaum and Chapman 1997, Hobbs and Harris 2001). Thus, as stated by Apfelbaum and Chapman (1997), “Success in restoration stems from being able to adapt to changing conditions just as functional ecosystems do.”

The process of measuring success stems from the project’s goals, objectives, and performance standards (Apfelbaum and Chapman 1997, Hobbs and Harris 2001, SER 2002, Hargrove et al. 2002). Measuring the success of restoration is challenging and the need for more effective measures of success is evident (White and Walker 1997, Mitsch and Gosselink 2000, Hobbs and Harris 2001). Specifically, there is a need for techniques that effectively compare attributes of the restored ecosystem to the reference conditions (Hobbs and Norton 1996).

The following list presents a number of strategies that have been put forth for evaluating the criteria of restoration:

1) Direct comparison of selected parameters from restored ecosystem and reference information: Because of the number of potential parameters, it is most effective to select a small set of traits that collectively describe the ecosystem and relate specifically to the goals of the project (SER 2002). Performance standards (the acceptable level of each attribute relative to the reference) should be established *a priori* (SER 2002). Several techniques have been developed to compare floristic composition of a degraded or restored ecosystem with reference conditions. The “coefficient of conservatism” is a means of characterizing the degree of fidelity of a species with undegraded, native habitat (Taft et al. 1997, cited in Allison 2002). On an 11-point scale, a score of zero indicates a weedy species found in any habitat and a score of ten indicates a species that is susceptible to degradation and restricted to native habitat. The “floristic quality index” is a measure that combines species richness with conservatism by multiplying mean conservatism values by the square root of species richness (Allison 2002). Similarity measures including “percent similarity” based on quantitative data and “coefficients of community” based on presence/absence data have been used to assess the similarity in species composition between stands. The former is more appropriate for measuring subtle changes over time, whereas the latter is more appropriate when the aim is to identify gross differences in composition (Westman 1991). Multivariate cluster analysis has been used to identify *a priori* reference communities for stream reaches with a specified morphology or landform and to thus characterize reference composition and structure for different community types (Harris 1999).
2) Comparison of parameters with natural range of variability: This approach has been used to assess forest condition (Walker and Boyer 1995, cited in Hobbs and Norton 1996) and could be an effective means of measuring restoration success (Hobbs and Norton 1996). Parameters for a range of key structural and compositional features are selected. For example, if a goal is to restore hydrological balance, parameters might be compared to a predetermined range of

runoff rates and changes in the water table. If the goal is to restore native species composition, parameters might be compared to the range of variability in the density of the dominant species and structural diversity measures (Hobbs and Norton 1996). Harris (1999) described how multivariate cluster analysis could be used to identify the natural range of variability for sets of parameters.

3) Attribute analysis: Quantitative and semi-quantitative measures can be used to determine the extent to which each goal is achieved (see attributes listed in "Attributes and performance standards" above (SER 2002)). Attributes may include, for example, "the restored ecosystem consists of indigenous species to the greatest practicable extent," "the restored ecosystem contains a characteristic assemblage of the species that occur in the reference ecosystem," "the restored ecosystem apparently functions normally for its ecological stage of development," "the restored ecosystem is sufficiently resilient," and "the restored ecosystem is suitably integrated into a larger ecological matrix or landscape." An approach similar to that used by McLachlan and Bazely (2001) to characterize the vulnerability of understory species based on their relative occurrence in reference stands compared with restored sites may be used to assess the extent to which the flora is characteristic of reference sites. Several estimates of a system's response to resilience measures have also been characterized (Berger 1991, Westman 1991, Hobbs and Norton 1996).

4) Trajectory analysis: A large set of data collected over time is plotted from the restoration site to determine if the site is on a developmental trajectory comparable to reference conditions (SER 2002). A technique for trajectory analysis has been developed by Minchin et al. (2001) to measure vegetation changes over time in order to ascertain which sites were on trajectories toward wetland composition. Minchin et al. (2001) used non-metric multidimensional scaling (NMDS) to reveal both trends in vegetation composition and vegetation dynamics. Stanturf et al. (2001) suggested that parameters such as soil carbon and soil bulk density could be measured over time for a restored site and compared with data from a chronosequence of sites with the same soil type.

Temporal Considerations

There are a number of considerations related to time that need to be considered when planning and conducting a restoration. First and foremost is the long time that it takes for an ecosystem to be restored. Jackson et al. (1995) suggested that 10-50 years should be sufficient time to demonstrate the success of a restoration project; within 10 years, evidence toward goals, for example tree seedling establishment, should be observable. Mitsch and Gosselink (2000) suggested that a longer time (perhaps a human lifetime) is needed to evaluate the successful restoration of forested wetlands. Mitsch and Gosselink stated that insufficient time is one of the major reasons for “failure” of wetland restorations. A considerable amount of time is necessary to ascertain if a restored ecosystem can sustain natural disturbances, exotic invasions, and cyclic phenomena (NRC 1992). A related issue is the long time that it takes to conduct experiments and publish results as a part of restoration studies (Michener 1997, Covington et al. 1999). It is also important to acknowledge the temporal limitations of data from reference sites; data represent a snapshot or a series of snapshots in time and must be used judiciously in light of this (White and Walker 1997). Hargrove et al. (2002) emphasized that criteria for success must be associated with a specific timeline for achievement.

Social, Cultural and Economic Considerations

Numerous authors have suggested that human values should be a primary consideration when establishing goals for ecological restoration and determining restoration success (Jackson et al. 1995, Apfelbaum and Chapman 1997, Covington et al. 1999). Sometimes social goals may be successful even if the criteria for ecological success are not met (Allison 2001).

It is appropriate for stakeholders to be involved in restoration planning and implementation and for restoration plans and progress to be reviewed by outside evaluators (Cairns 1989, NRC 1992, Clewell et al. 2000). Moreover, public landowners are often a key to restoration (Ogren 2002). Restoration can be an accessible means of educating the public about conservation and ecology (Hobbs and Norton 1996, Allison 2002) and it is sometimes possible for volunteers with little experience to collect meaningful data (Bader 2001). Higgs (1997, cited in Hobbs and Harris 2001) wrote that "good ecological restoration entails negotiating the best possible outcome for a specific site based on ecological knowledge and the diverse perspectives of interested stakeholders: to this end it is as much process as product oriented." Using economic resources effectively is critical and an analysis of costs (including the incremental costs of various actions (NRC 1992)), benefits, risks of failure, and environmental impacts can help evaluate various restoration options (FISRWG 1998).

Recommendations and Guidelines

Included in Appendix B are summaries of restoration planning guidelines from a variety of sources. The guidelines are useful for determining what steps to take in planning and implementing restoration and for providing a conceptual framework for the restoration.

Although the guidelines reflect a variety of different emphases, the frameworks have a number of features in common. Important steps in ecological restoration (not necessarily in the order they should be accomplished) include:

1. Identify stressors and the ways that structure and processes have been degraded.
2. Assemble ecological information from multiple sources to guide restoration.
3. Plan carefully before implementing a project.
4. Establish reasonable and achievable goals that can be measured.
5. Generate and test hypotheses.
6. Use information from experimentation and monitoring to adapt the restoration plan and goals.
7. Carefully monitor and document parameters that help measure progress toward goals.
8. Share results with the scientific community.
9. Plan restoration at the landscape scale.
10. Reflect societal values in setting goals.

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4. RESTORATION TECHNIQUES

This section deals with the mechanics of forest restoration, including planting and site preparation methods, planting materials and nursery protocols, weed control, herbivore control, and fertilization. There is an abundance of quantitative and qualitative information regarding oak species, especially the southeastern oaks but also northern red oak (*Quercus rubra*) and white oak (*Q. alba*), and very little (mostly qualitative) information regarding other species of deciduous trees. Some of the information is directly related to reforestation of fields, while some of it, especially regarding northern red oak, comes from a body of studies related to oak regeneration following timber harvest; also, some useful information is derived from timber plantation plantings. Regardless of the reforestation or post-harvest regeneration focus of the studies or projects, a great deal of the information is applicable to forest ecosystem restoration, as early-year survival and growth of trees are seen to be the important factors in all cases where trees are regrown on abandoned fields or in logged forests. The first one to three years of growth are of greatest importance, and many of the differences related to planting technique and stock age are not significant to the growth of trees beyond the third year after planting (Zaczek et al. 1997, Weigel 1999). In addition to information regarding trees, separate subsections address restoration concerns related to soils, herbaceous plants, and animals.

Direct-Seeding

Both planting of seedlings and direct seeding are effective means of restoring trees to old-fields (Johnson and Krinard 1985b, Allen 1990, Bullard et al. 1992, Harmer and Kerr 1995, Smiles and Dawson 1995, Cornett and Evenson 1999, Isenbrands 1999, Allen et al. 2001, Peterson pers. com.). Almost all of the quantitative work uncovered in the literature, however, relates to oaks. Nevertheless, the Big Woods Project in Minnesota has also been direct seeding other genera (Cornett and Evenson 1999), and Peterson (pers. com.) reported that he has increasingly relied on direct seeding native tree species mixtures. There is a long history of directly seeding trees in fields dating back to the 15th or 16th century in England where acorns and also ash samaras were planted in dibble holes or along with or shortly after planting of a cereal crop (Harmer and Kerr 1995); the history of planting may be as long in the area of present-day Germany as well.

The advantages and disadvantages of direct seeding have been summarized by Allen et al. (2001) and Bullard et al. (1992). They state the following advantages: flexibility in timing of planting seeds, ability to plant seeds faster than seedlings, provision for natural processes to select for establishment and superior competitiveness of the new trees (in relation to micro-site differences), no disturbance of roots such as occurs when seedlings are removed from nurseries, and substantially lower cost. Bullard et al. calculated that in the Southeastern U.S. direct seeding was approximately one-third the cost of planting nursery stock ($57-70 vs. $160 per acre, including disking or brush-hogging, planting and cost of planting materials in 1992). Evenson (pers. com.) calculated that direct-seeding was 45% the cost of planting seedlings ($740 vs. $1640 per acre), but seed collection for his projects has been done by very low-paid conservation corps crews or community-service crews.

Among the disadvantages listed were the need for a site greater than 2.5 acres to decrease rodent damage to acorns, differential success in germination and establishment by species, proven reliability only for oaks and some other large seeded species (although see Cornett and Evenson 1999), slower establishment and development (although long-term growth may not differ from that of planted seedlings), a potential that in any given year success by direct seeding may be somewhat less than by planting (an untested statement, but acknowledged also by Evenson (pers. com.)), and greater difficulty in monitoring since it is hard to find germinated seedlings for several years. Allen (1997) contended that direct seeding allows for better prospects of natural establishment of species into planted land, because naturally arriving species and planted species were observed to be more similar in height; also direct-seeded plantings were much more patchily stocked and thus provided more space for natural establishment. He did note, however, that prospects for natural establishment were fewer in isolated fields and that heavy-seeded species arrived more slowly.

Bullard et al. (1992) noted that there is generally greater success in direct seeding in the red oak group than in the white oak group (uncited information), although germination/survival percentages reported by several authors do not appear to support that difference (Sluder 1965, Johnson and Krinard 1985a, Lockhart et al. 1995). Differences in direct seeding of red versus white oaks could be of significance for restoration in Champlain Valley clayplain and floodplain forests, where there is one species in the red oak group and three in the white oak group; also of significance is that white oaks generally show more population-level periodicity (masting) than do red oaks, and thus white oak acorns may not be as available any given year (Johnson et al. 2002).

General early direct-seeding guidelines (Schlich 1904. cited in Harmer and Kerr 1995) included the following: species with large seeds are better suited to the method than those with delicate seeds, good test-proven germination rates are necessary, covering seeds with soil reduces predation and protects against desiccation, sufficient amount should be sown (Harmer and Kerr note that 100,000 seeds per ha (variable by species) was recommended by Evans (1984)). Allen et al. (2001) provided diagrams and illustrations of acorn planting machines used in the southeast.

Specific acorn planting guidelines have been developed through empirical testing. Acorns are best planted at a depth of 5 cm or less (Sluder 1965, Johnson and Krinard 1985b, Lockhart et al. 1995, Smiles and Dawson 1995, Tomlinson et al. 1997). Deeper planting has a negative impact on shoot:root ratios, which are an important measure of seedling survival and competitiveness the first few years of growth (Tomlinson et al. 1997). Lockhart et al. , however, found greater emergence of white oak acorns that were placed on either the soil surface between the organic and mineral horizons or beneath the litter and atop the fermentation (Oe?) layer.

Reported survival rates of direct-seeded acorns are 43-91% (Johnson 1981, Johnson and Krinard 1985b, Kolb et al. 1989, Lockhart et al. 1995, Stanturf 1995), with lower rates reported when acorns are planted below 5 cm (Sluder 1965, Johnson 1981, Johnson and Krinard 1985b). One study on clay soil (Sharkey series) in a Lower Mississippi Valley flood-prone site reported direct-seeding first-year survival of only 30% (Williams and Craft 1997). Direct-seeding of white ash (*Fraxinus americana*) and white pine (*Pinus strobus*) in the Midwestern U.S. yielded germination rates of 11-32% and about 50%, respectively (Kolb et al. 1989).

Acorn size does not appear to be of great importance, but there are few studies to rely on. Auchmoody et al. (1994) found that red oak acorn size was not a significant factor related to insect or small mammal predation or seedling survival after three years, although larger acorns did show slightly greater height growth. In another study of red, pin, and black oaks, researchers observed a higher percent of seedling emergence in larger acorns (Smiles and Dawson 1995).

A factor that must be considered in direct seeding is acorn predation. Gribko (1995) found that *Contrachelus* sp. weevils damaged surface-sown red oak acorns to a greater extent than those sown 2.5 cm deep, but much of the insect damage was not severe enough to cause lack of seedling success. Galford et al. (1991), however, obtained contrary results that insects did affect establishment. Small mammals also prey on acorns, and surface-sown acorns may be nearly totally wiped out by predators (Sluder 1965, Auchmoody et al. 1994). Auchmoody et al. observed about 50% acorn loss to predators, for acorns sown at 2.5 cm depth. Sluder found that screen protection over acorns planted 5 cm deep increased establishment by 23%. The minimum field size of 2.5 acres to reduce small mammal predation of acorns (Bullard et al. 1992) comes from the southeastern U.S. and it is not known if this same measure is relevant to the northeast; it is likely that landscape position of a field in relation to forest patches and wooded fencerows would have some affect on small mammal population size and feeding in fields planted in acorns or other seeds.

The Big Woods Project restorationists have been using a two-stage direct seeding protocol (Peterson pers. com.). First, heavy seeds (oak (*Quercus* spp.), hickory (*Carya* spp.), walnut (*Juglans* spp.)) are broadcast, often using a fertilizer spreader, a sand spreader (as used to spread road sand and salt) or hand-sown (Cornett and Evenson 1999, Evenson pers. com., Peterson pers.

com.); these are lightly disked 1-2” into the soil. Second, light-seeded species (maple (*Acer* spp.), ash (*Fraxinus* spp.)) are broadcast and either left on the surface or dragged into the soil with a chain. Mixing the seed with sand can aid in the mechanical dispersal of light seeds (Evenson pers. com.). All of this has normally been done in the fall, following site preparation for at least one growing season. Cornett and Evenson (1999) reported that early and mid-successional species established the most successfully; species with poor success were bitternut hickory (*Carya cordiformis*), sugar maple (*Acer saccharum*), basswood (*Tilia americana*), American hazel (*Corylus americana*), and several fleshy fruited species (eastern red cedar (*Juniperus virginiana*), cherries (*Prunus* spp.), viburnums (*Viburnum* spp.), and prickly-ash (*Zanthoxylum americanum*)). Lack of seed scarification may have been a factor in the poor success of fleshy-fruited species. The authors noted that one gallon per acre yielded the following mean number of seedlings per acre: American elm (*Ulmus americana*)—588, green ash (*Fraxinus pennsylvanica*)—600, oak (*Quercus* spp.)—106, dogwoods (*Cornus* spp.)—420.

Cornett and Evenson (1999) provided five recommendations: 1) focus collection efforts on easy-to-grow species; 2) intensify the collection efforts and seeding of species that had been seeded at a relatively small number of sites, or in relatively small quantities, to determine their potential for success; 3) experiment with different post-seeding weed-control methods to clarify the stage at which failure occurs for some species; 4) consider abandoning collection efforts for fleshy fruit-bearing species, unless an alternative seed treatment or weed control technique improves establishment; 5) avoid planting box-elder (*Acer negundo*) because of its invasive tendencies. The fifth recommendation should receive some scrutiny, because box-elder is a native tree of floodplain forests in Vermont, and may be useful for restoring areas where establishment of other species has proven difficult. Also, abandoning collection of fleshy fruits may not be as appropriate as abandoning direct seeding of fleshy fruits; at any rate, it appears that more work is needed to determine protocols for direct seeding of drupes. Prickly-ash can be dangerous to collect, as the oil in the fruit causes extreme sensitization of the skin to solar rays;

in Minnesota several collectors were hospitalized due to severe sunburns after collecting prickly-ash seed (Evenson pers. com.).

Seedling Stock

Seedling age and nursery propagation techniques have been shown to be significant to the growth of oak seedlings; few papers report on important seedling characteristics for other eastern hardwood trees. Root structure, particularly a greater number of first-order lateral roots (FOLR) is one of the most important parameters for outplanting success of red oak and black walnut (Ponder 1997, Isenbrands 1999, Kormanik et al. 1999). Root collar diameter (RCD) is another important parameter (Isenbrands 1999, Clark et al. 2000). Clark et al. noted that 40-60% of nursery stock was of unacceptable size for planting (and they cited three other studies that confirmed the same), but they did find that seedlings could be graded visually into premium, good, and cull grades.

Both bare-root and container stock can be used in successful plantings. Cost, ease of planting, and likelihood of improper handling techniques are considerations. Container stock may better tolerate poor handling and being planted outside the normal planting season (Kerr 1994), but not all researchers agree with the latter assertion regarding season of planting (Hodge 1991). Container stock is more costly to grow, and it may be more difficult to plant (Zaczek et al. 1997). Larger, older container stock is definitely more difficult to plant in clay soils (Lapin pers. obs. from planting 2-3”-tube stock in the fall of its third year), and it also appears to be highly susceptible to frost-heaving in clay soils (McQuilken 1946, Stroupe and Williams 1999). Site preparation that removes the vegetation cover, which slows cooling and adds stability to soil structure, may exacerbate heaving; bare-root stock and acorns have also been observed to heave in heavy soils (Stroupe and Williams 1999). For flood-prone sites, Nuttall oak (*Quercus nuttallii*) container stock appeared to be a better performer in the Lower Mississippi Valley (Williams and Craft 1997).

Brissette et al. (1991) provide a wealth of information about container stock; although it specifically references experiences with the southern pines (*Pinus* spp.), their chapter is a useful starting point for learning about container stock. Among the points necessary to mention, they noted that root malformation, especially root spiraling in clay soils, may occur if seedlings are in the containers longer than the optimal time (optimal time will differ by species and container size). Also, extractable plugs (seedlings intended for removal from the container before planting, as opposed to container types in which the container is put in the ground with the tree) are not well suited to machine planting, but tube and block type container stock is. They noted that in clay soils removing a soil core the same size and shape as the plug (as opposed to dibble-stick or bar planting) has resulted in better performance in those heavy or compacted soils; nevertheless, planting tools used in clay soils must be designed to avoid soil compaction or case-hardening of the hole's sidewalls. Brissette et al. also noted that depth of planting is more critical with container stock than with bare-root stock, for if the plug – the growing medium and the roots within it – is not fully covered with soil, the roots can dry out rapidly through wicking of moisture from the growing medium into the air. Furthermore, when containers with small soil volumes are used, timing and application of irrigation and fertilization are critical. The chapter also elaborates upon planting techniques for container stock.

After six years study of observing red oak grown from 1-0[1] and 2-0 bare-root stock, 2-0 container stock, and direct seeding, Zaczek et al. (1997) concluded that the marginal increase in height growth of container stock was not worth the extra cost and difficulty of planting. They suggested planting 2-0 bare-root seedlings that were top-clipped and under-cut in the nursery; they mentioned that clipping the tops and trimming the roots resulted in only slight benefits, and could be foregone. 1-0 seedlings were not recommended, as they only slightly outperformed direct seeding and were therefore not thought to be worth the cost or effort. Researchers in

[1] Nursery stock age is presented in the format of (# of years grown in first bed)-(# of years grown in transplant bed). Therefore a 1-0 seedling was grown for one year with no transplant, a 2-0 seedling was grown for 2 years with no transplant, and a 1-1 seedling was grown for two years and was transplanted to another bed for the second growing season. Transplanting influences root growth and root:shoot ratio.

Ontario found that 1+1 red oak seedlings outperformed 1-0 stock in all treatments, regardless of whether seedlings had been top-clipped or browsed (Gordon et al. 1995).

Growth differences in top-clipped and unclipped oaks have been observed in the first two to three years after outplanting; by year four heights and growth rates did not differ (Zaczek et al. 1997, Weigel 1999). Weigel, however, did observe that red and white oak differed in response to top-clipping and undercutting. Red oak that had undergone neither treatment outperformed all other treatments after 13 years, whereas white oak that had neither treatment showed significantly poorer performance than other treatments. Weigel reported that seedlings that were not undercut or top-clipped had the lowest survival. Hence, with regard to top-clipping and undercutting there may be a trade-off between initial survival and later growth for some species. There may also be differences related to age at which seedlings are planted.

Among the southeastern bottomland oaks two studies detected conflicting results. Mullins et al. (1997) found no difference in height after five years of growth of cherrybark oak (*Quercus pagoda*) grown from container, bare-root, or direct-seeding, whereas Ozalp et al. (1997) found that bare-root stock exceeded directly seeded oaks (cherrybark, Nuttall, Shumard (*Quercus shumardii*), and water(*Q. nigra*)) after five years. In the latter study, soil series, including the Mississippi bottomland Sharkey clay (similar in particle size proportions to the Champlain Valley clay soils), did not have an effect on survival or growth. Hence, it must be concluded at this time that there are no clearly evident advantages to seedlings propagated by different techniques, that results have varied by species and research project, and that good success has been achieved with nearly all the commonly used methods.

Survival of bare-root stock has generally been 50-90%; higher percentages are related to weed-control practices which are discussed below (George et al. 1991, Rink and Kung 1991, Ozalp et al. 1997, Ponder 1997, Weigel 1999). Rink and Kung (1991) found survival of white ash 1-0 seedlings to vary from 50-97% in various combinations of provenance and weed control practices.

Nursery protocols can have significant effects on seedling quality, as measured by FOLR, RCD, shoot:root ratios, etc. As in the field, depth of acorn planting is important, and planting red oak acorns deeper than 5 cm resulted in an unfavorably high shoot:root ratio (Tomlinson et al. 1997). The same authors found that hardwood sawdust mulch (aged two years) had a negative effect on FOLR development, but corncob mulch did not. Planting density in the nursery bed influenced root growth and dry mass of the total seedling (Tomlinson et al. 1997), but for red oak in the southeast, Howell and Harrington (1997) found no survival, diameter or height differences after two years among bare-root seedlings grown at different densities in the nursery. They, therefore, recommended growing smaller seedlings at higher densities and using faster planting methods (including root-pruning in the field) to maximize efficiency; those recommendations appear to conflict with suggestions to use 2-year old stock (Gordon et al. 1995, Zaczek et al. 1997).

A method recommended for maximizing efficiency for planting cherrybark oak was to grow seedlings in large containers (11.5 cm diameter), fertilize in the nursery, remove them from the containers, and remove soil from the roots (basically create a bare-root plant) prior to shipping and planting (Howell and Harrington 2002). Small and medium container stock (3.5 and 6.5 cm diameter pots) were found to be optimum for cost efficiency, but the biological advantages of the larger stock were noted and the soil-removal technique allowed the larger stock to be more economically competitive (by reducing shipping costs and planting costs/difficulty). Most of the available data regarding response to nursery conditions and planting size refer to oaks in the red oak group. Little has been documented regarding species in the white oak group, and even less regarding other eastern hardwood genera.

Big Woods restorationists have not been concerned with grading seedlings to get maximum early growth rates. Rather, by not culling the smaller, weaker-appearing seedlings (i.e., planting the nursery run) they are planting for maximum genotypic heterogeneity and thus maximum phenotypic variability (Evenson pers. com.).

When considering how to maximize efficiency (both planting success and post-planting treatments) for the various species, ecoregional and landform/soils differences should not be discounted as significant factors. Weigel (1999) found that planting success and growth parameters were shown to differ by ecoregion. He attributed the differences in red oak plantings in two Midwestern U.S. ecoregions to competition with faster growing trees in the more fertile region. Thus, controlling tree competition may be more important to the success of restoring a mix of native species in some regions than in others. It is important to keep in mind that studies are not necessarily 100% transferable from one region to another, and within the same region, and even the same landform, differences by species have been documented.

Natural Succession

The literature on natural succession is vast and its review was not in the scope of the current project. Natural succession is an accepted technique in the restoration toolbox, and is the one that is the least intensive, and thus least expensive, to implement (Brown and Lugo 1994, Harmer and Kerr 1995). Succession does have greater unpredictability, however, and it may take a longer period to achieve a woodland canopy compared to planting (Harmer and Kerr 1995). Harmer and Kerr also stated that natural regeneration of forests is often more difficult on moisture retentive, heavy, fertile soils, where there is greater competition with luxuriant field vegetation. It is unknown at this time whether that assessment would hold true for the moist and fertile clayplain soils in the Champlain Valley, but it does not appear to hold true for the floodplain soils, which are light-textured and fertile.

Availability of and distance to seed source are two of the most important factors in the rate and trajectory of natural succession. In the Lower Mississippi Alluvial Valley, distance to seed source, dispersal mode, and elevation (highly correlated to soil type and flooding regime and thus a major factor in the ecosystem/natural community types in that area) strongly influenced the pattern and rate of secondary succession (Battaglia et al. 2002). Pan et al.'s (2001) St. Lawrence Valley work showed that past land use (cropland versus pasture) had an effect on the secondary forest composition, and that the effect was stronger on clay deposits than on morainal deposits; coniferous forest was more abundant on former pasture, and the relationship was especially strong on the clay soils.

Rates of tree succession into abandoned fields have been measured in various places. Some data from relevant examples are Harrison and Werner's (1984) Michigan study that found oak forest required 15 years to establish in an old-field, where a seed source was 300-400 m distant. Robertson (1992)measured a maximum dispersal distance of about 30 m in 24 years for bur oak establishment into a field adjacent to a woods; 60 seedlings had established in the 4.7 acre field in 30 years, and the dispersal distances suggested that mammals, not birds, had been the primary dispersers. In southern Ontario, a field surrounded on three sides by woods had only four tree species established as large trees 50 years after field abandonment; a total of 12 tree species were present in the field (Maycock and Guzikowa 1984). Allen et al. observed the relative paucity of heavy-seeded species (oaks, hickories) in natural succession after two years of field abandonment in the Lower Mississippi Valley, and an extension forester in Ontario had mentioned several years ago that in their lake plain region, oaks did not readily spread into the abandoned fields (Boysen pers. com.). Natural oak establishment in the southern Champlain Valley's old fields does appear to occur close to seed sources, but the establishment rates (distances, abundances) have not been studied.

In addition to unpredictability of tree establishment rates, natural succession typically proceeds in a non-random but patchy spatial pattern. Battaglia et al.'s (2002) study showed different patterns of clustering and dispersion for different tree species. Site location in relation to seed source and prevailing winds can be significant, as more seedlings are commonly seen on downwind sites (Allen et al. 1998). The patchiness can be put to advantage in a program of natural succession enrichment plantings that could build upon and help to accelerate natural succession by enriching the site with species that are not naturally establishing.

In a comparison of three Kentucky sites – a woods restored by planting, a site of natural succession, and a mature forest remnant – it was found that although the structure of the stands were similar after 50 years of forest regrowth (or were on trajectories that indicated future similarity), there were differences in species composition (Shear et al. 1996). Oaks and hickories were under-represented in the secondary forests. The time frame of this research seems to have been very short, and it is not surprising that there are species composition differences after only 50 years of regrowth.

Some authors recommend using natural succession as a tool for advancing or accelerating forest restoration (Lugo 1988, Allen et al. 1998, Brown and Amacher 1999). If necessary, amelioration of limiting conditions (“stressors” per Yates and Hobbs (1997)) can be used to facilitate the natural invasion, establishment, and colonization of many native species. Planting early successional species, or allowing their natural establishment, can play an important role in ecosystem development, as they are important in “initiating the interactive phases of succession among the biological, physical, and chemical components of the environment” (Brown and Amacher 1999). Along these lines, it is also important to note “assembly rules theory,” which is based on experimental evidence that the first arriving species can have lasting impacts on the future composition (and thus structure?) of an ecosystem (Lockwood 1997). Assembly rules theory appears to have been derived principally from studies of aquatic ecosystems, and it was beyond the scope of this review to look into that literature in any depth.

For a small amount of information on local, Champlain Valley succession patterns please refer to Graves’ (2001) data from successional clayplain forests in the Hubbardton River valley. Also, Barbara Otsuka, an Antioch-New England Graduate School student has embarked upon a study of succession on clay soils in the southern Champlain Valley; she should have a report available by 2005.

Provenance and Genetics

Researchers have been studying growth differences among seedlings of trees from different populations of the species since 1745, and there is much evidence that numerous ecological differences among individuals and populations are genetically based (Barnes et al. 1998).

Seedling quality, therefore, is in part determined by genotype. In an investigation of selection for superior timber trees by phenotype it was seen that trees rated as “superior” did not throw progeny that had advantages (growth and form) for reforestation; the authors thus recommended selecting acorns from a wider selection of “above-average” trees rather than from fewer “superior” trees (Stringer et al. 1995). Results from half-sib tests indicated that FOLR development and number of competitive seedlings were heritable (Kormanik et al. 1999). The implication is that seed collected would need to be tested in the nursery prior to selecting the “better” parent trees. Provenance was also observed to be significant for height and diameter growth of white ash regenerated from 1-0 seedlings (Rink and Kung 1991). For projects in which restoration of a mix of native species is an objective, and timber quality is not of concern, parent-tree selection may not be necessary. A strategy more efficient than testing for trees that have higher out-planting or direct-seeding survival may be to collect seed from a wide selection of parent trees. That would also benefit the restoration by starting with a more diverse rather than more similar gene pool, so as to not set the stage for founder’s effects and other deleterious genetic impacts of small, isolated populations (Templeton et al. 1990) later in time.

In The Big Woods Restoration local seed source is preferred (Peterson pers. com.). Some partners have a strict definition of local origin that defines local to be within 23 miles of the restoration site, but that is often impractical and a radius of 60-90 miles has been necessary at times.

Site Preparation and Control of Vegetation Competition

Various methods are used to control inter-specific competition; the most common techniques are field disking, herbicide application, and brush mats (commercially available porous fiber mats), used singly or in combination. Of primary importance is “weed” control the first two to three years after planting (von Althen 1981, Godel 1982, Rink and Kung 1991, Demchik and Sharpe 1999, Isenbrands 1999, Peterson pers. com.). McLeod et al. (2000), however, were surprised to find that among oaks, water tupelo (*Nyssa aquatica*) and bald cypress (*Taxodium* *distichum*) restored to a field on the South Carolina coastal plain, after three years of growth only the latter species was significantly shorter in the control treatment (herbicide and mowing were the experimental weed-control treatments in the study); their results may be atypical. Many restorationists have used glyphosate to reduce plant competition (Kolb et al. 1989, Ponder 1997); it is often used both prior to planting and in years subsequent to planting, and the literature strongly indicated that band applications are the norm.

Mechanical cultivation, usually disking, is also often used to clean fields prior to restoration plantings and to control weeds after planting (Krinard and Kennedy 1983, Morris and Lowery 1988, Allen et al. 1998, Peterson pers. com.). Disking two or more times during the first growing season in southern pine plantations was recommended by Minogue et al. (1991), who noted that one can employ either a linear pattern or a more intensive two-way pattern where planting spaces are wide and regular. Although weed control is one of the most important factors for success of planted seedlings (Demchik and Sharpe 1999, Peterson pers. com.), it should not be assumed that cleaning a field is fully advantageous, for Allen et al. (1998) found that although disking usually enhanced survival and growth of planted seed or seedlings, it may temporarily reduce the natural establishment of other seedlings, especially on wetter sites. Their hypothesis was that large soil clods and a soil surface that is perhaps more prone to drying inhibited establishment of seed naturally dispersed to the area.

In a study of direct-seeded red oak, white ash, white pine and yellow-poplar (*Liriodendron tulipifera*), Kolb et al. (1989) concluded that fern and grass presence had a slight beneficial effect on the germination of red oak (75-88% germination weed-free vs. 86-91% control), and to a lesser extent of white pine, but there was not a beneficial effect for white ash or yellow-poplar). Weed-control, however, was strongly related to greater first-year diameter growth for all four species, and to increased height growth of all species except white pine.

Mowing is another method used for weed control. On Sharkey clay soil in the Southeast, mowing has been observed to be inferior to disking for weed control (Krinard and Kennedy 1981, 1983). The authors noted (1981) that disking enhanced nitrogen availability due to both incorporation of green manure into the soil and increasing mineralization rates (but note in soils discussion below that forest typically has lowered nitrogen availability and greater immobilization of nitrogen, compared to agricultural fields); water infiltration and gas exchange with the atmosphere are also enhanced by disking, as is root proliferation. Morris and Lowery (1988) compared effects of three common mechanical site preparation techniques: disking, bedding and subsoiling. Disking alters the pore structure of soil and therefore decreases resistance to root penetration and increases water and air transport in the soil; thus root density is increased. Bedding, in addition to the above, raises soil above the water table and rooting density has been shown to increase in both sandy and clayey soils. Subsoiling neither increases the volume of large soil pores nor reduces mechanical impedance to root penetration; it may lead to a more uniform distribution of roots, but there have been few studies and none with hardwood species were reported. All three techniques will increase water infiltration on compacted or puddle-prone soils, and will improve available soil moisture by decreasing weed competition and increasing root volume. It should be noted that increasing water infiltration must not be seen as a benefit to ecological restoration in ecosystems where soils are naturally dense and infiltration is naturally slow; however, where excessive compaction has occurred due to prior land uses, it may be advantageous to improve infiltration to more “natural” levels.

von Althen (1981) found that on clay loam soils on the Lake Ontario lake plain, the best site preparation and weed control treatment for establishing white ash, black walnut (*Juglans nigra*), silver maple (*Acer saccharinum*), basswood, and hybrid poplar (*Populus* sp.) to old fields was an initial summer plowing and disking followed by one or two more diskings in the fall prior to planting seedlings. If quackgrass *(Agropyron repens)* was dense, he recommended herbicide application of pronamide (Kerb®, @ 2.2kg/ha). All plots had post-planting weed control via applications of Princep® Nine-T (simazine). Poorest growth was in plots with no site preparation, and also in plots with paraquat application prior to planting. Planting in furrows of plowed but not disked fields gave the worst height growth, while planting in plowed strips treated with simazine yielded better growth than planting in individual planting circles treated with simazine. von Althen's report included appendices that help with calculating herbicide dosage of Princep® and Roundup® for spot treatment of different size spots.

Peterson (pers. com.) has found that on agricultural sites in the Big Woods Region, weeds take over if no weed control is practiced. He relies mostly on herbicides for weed control, but has also used mowing, brush mats, and mulch. Most of the plantings in that project are on private lands and Peterson wrote, "Maintenance is not as glamorous as planting and so it is harder to get landowners to do all the follow-up required for a planting." Another Big Woods restorationist, Evenson (pers. com.) has not been practicing post-planting weed control; he has been planting at much greater density and using 50% shrub composition to quickly establish a canopy that will shade out the grasses.

Brush mats are an often used method for control of vegetation competition (Van Sambeck et al. 1995, Derleth pers. com., Peterson pers. com.). They can be used with or without prior site preparation. Van Sambeck's experiment with white ash and silver maple first-year seedlings showed that although several different weed control techniques all yielded greater height growth than no weed control, seedlings in the plastic films treatments were taller than any others. He used both solid plastics (opaque and black) and porous black mats; the latter allowed for moisture penetration and more heat exchange, while solid black plastic resulted in the highest daily change in soil temperature. The solid plastics began to disintegrate after two growing seasons, whereas the porous material was thicker and remained intact through the third season (at which time the experiment apparently ended). Van Sambeck's site had been plowed and cultivated prior to planting. The documented growth advantages of using brush mats for weed control need to be weighed against their cost in comparison to other methods. That cost calculation should include either retrieving mats after two to four years, or allowing them to slowly deteriorate and litter the landscape. It is noteworthy that in one Big Woods restoration research project, use of brush mats without tree shelters led to increased deer (*Odocoileus virginianus*) browsing and greater rates of mortality, compared to unsheltered trees with no brush mats; the brush mats, it seems, acted like magnets or flags to alert the deer to a good meal of red oak (Stange and Shea 1998).

Woodchip mulch, in conjunction with application of 12-12-12 granular fertilizer, has been an effective method in white oak plantings in Illinois (Schaal 1996). Schaal mulched a 30 cm diameter area with 8-10 cm of woodchips, over a direct-seeded acorn. Mulch and fertilizer were reapplied for three years. Schaal's note anecdotally reported growth of these white oaks was four times greater than mean growth in four years; it is unclear whether the mean he compared to was from a control planting on the same site, or from another information source.

Fire is another weed-control treatment that can be applied, but more often planned burns are instated to restore an altered or missing ecosystem disturbance. In restoring lawn sites to oak savanna on the Indiana dunes, it was found that fire was ineffective for eradicating Kentucky bluegrass (*Poa pratensis*) and quackgrass in moist fertile areas where topsoil had been imported, despite reports from other literature that fire was useful for eliminating those grasses (Choi and Pavlovic 1998). Choi and Pavlovic observed that fire only removed the above ground plant parts, and thus the grasses readily regrew; herbicide and sod removal were found to be equally effective methods for removing the exotic lawn grasses.

Lowery and Gjerstad (1991) gave detailed information regarding site preparation methods for southern pine plantations; their chapter provided useful tables regarding herbicides registered for such uses and application method and timing information. They also developed an informative table to present advantages and disadvantages of mechanical and chemical site preparation techniques. Allen et al. (2001) also provided tables of commonly used herbicides, the weed species susceptible to sulfometuron methyl (Oust), and timing guidelines (for Southeast U.S.) for various herbicides.

Seedling Protection

Herbivore damage to seedlings is a major concern. Deer, rabbit (*Sylvilagus* spp., *Lepus* spp.), voles (*Microtus* spp.) and mice (*Peromyscus* spp.) all browse on tree shoots and/or bark. Tree shelters provide protection from herbivores while seedlings are young, small, and most vulnerable, and there is extensive literature on the effectiveness of shelters for both preventing browse and for promoting survival and growth. Chemical repellants are another method used to ameliorate deer browsing.

Seedling mortality has been shown to be lower for numerous species when tree shelters are used (Lantagne 1995, Ward and Stephens 1995, Kinkead et al. 2002, Sweeney et al. 2002). Seedling growth in the first two to three years has also been shown to be significantly greater for sheltered seedlings (Lantagne 1995, Walters 1995, Ward and Stephens 1995, Mullins et al. 1997, Ponder 1997, Stange and Shea 1998, Clatterbuck 1999, Kinkead et al. 2002, Sweeney et al. 2002). Ward and Stephens (1995) noted, however, that for the conifers in their study (eastern hemlock (*Tsuga canadensis*) and Norway spruce (*Picea abies*)) the height increase was small and shelters may not be suitable or recommended. Interestingly, Walters (1995) did not find a significant growth advantage of sheltered versus unsheltered naturally regenerated red oaks; he did find that planted acorns survived in shelters, but not in chicken wire fences or without protection. It is clear from the literature that the height-growth advantage conferred by the shelters in the early years does not continue past year three or so (Lantagne 1995, Mullins et al. 1997, Clatterbuck 1999).

Although they provide excellent protection from herbivory, tree shelters do not come without some problems. Expense is one issue, and labor required to install them is another. Of greater concern to seedling growth is that without maintenance tree shelters can be detrimental to seedlings (Zaczek et al. 1997, Evenson pers. com.). Among the problems are tipping over, bird and insect nests inside shelters, and delaying of hardening for winter dormancy. Tipped shelters will either cause trees to grow at extreme angles or will smother them. Bird and wasp nests can cause leaders to curl and twist and thus deform trees. In solid tube style shelters the altered microclimate (elevated levels of heat and carbon dioxide) can allow seedlings to photosynthesize more, but can also lead to delay of dormancy, which can cause dieback. Evenson (pers. com.) contacted a shelter manufacturer and was told the lifting the shelter two inches off the ground will ameliorate that problem; clearly that is a very labor intensive solution. Net-type shelters that protect from deer herbivory but do not elevate temperatures around the seedling may be a better solution if delayed hardening off is a serious problem, but the net types will not protect from rodent herbivory.

Two other alternatives used for deer-herbivory protection are animal repellants and fencing. Five- to six-foot high electric fences are effective protection (George et al. 1991, Opperman and Merenlender 2000, Evenson pers. com.). Despite the seemingly high cost of fencing, it should not be discounted, as it may be a small investment relative to the entire restoration costs; furthermore, plantings can be designed to minimize perimeter:area ratios to cut down on fencing costs (Evenson pers. com.). Fences do become less effective over time, for deer begin to get inside, but seedlings greatly benefit from the protection in the crucial first years of growth (George et al. 1991).

Repellants that are used include commercial products such as Deer-Away, Big Game Repellant ® (BGR), HinderDeer, Ropel™, Rabbit Repellant ®, and coyote urine, as well as "home remedies" such as bars of soap and egg mixtures. BGR® contains putrescent whole egg solids and has been found to be among the most effective repellants (Swihart and Conover 1990, Andeldt et al. 1991). Andeldt et al. also found that fresh chicken egg solution and coyote urine performed similarly to BGR®. In their study, some repellency remained even after heavy rainfall. Soap bars hanging from apple trees, and even empty soap wrappers have been shown to be somewhat repellant, and different brands performed differently (Scanlon et al. 1987, Swihart and Conover 1990). Andeldt et al. (1991) caution that repellency effectiveness is a function of deer hunger and browse palatability, and that if deer are moderately hungry, repellants are unlikely to deter browsing.

In developing plans for herbivory protection, it is useful to note spatial differences in browsing patterns within a field. Higher levels of browsing and other deer activity occur closer to woods edges (Wester and Young 1997, Stange and Shea 1998). Therefore, either affording more protection to seedlings planted nearer to woods edges, or avoiding planting the most palatable species near the edges would appear to be efficient strategies.

Planting Methods

For detailed information regarding the mechanics of planting, including machinery and hand tools, techniques, acceptable weather and soil moisture conditions, and even terms for contractors, see Long (1991). The information is geared specifically for southern pine plantations, but the material is applicable to planting of any seedlings.

In large-scale restorations, tree or seed planting can be done by machine (von Althen 1981, Booth 1985, Anonymous 1994, Peterson pers. com.). Machine-planting is faster but results in plantation-type plantings (Allen 1997, Peterson pers. com.), and it may be less exact in placement of seedlings in relation to the root collar and stem-orientation (Long 1991). In clay soils, machine planting of seedlings may be problematic, because the planting slits tend to open due to soil drying and shrinking (von Althen 1977, 1981). von Althen, who worked on the Ontario clay plain, found that disking fields helped to lessen the problem; fields that were plowed but not disked had high rates of seedling death and depredation by mice due to open planting slits.

Different problems are apparently encountered in different soils, however. It seems that in some soils grass roots aid in mechanical planting, and therefore baring the soil and cultivating for more than one season can be problematic (Evenson pers. com.). In Evenson’s experience machine planting in soil that has no roots to bind it is difficult, because the planting slits fall open too loosely and do not hold together once the seedling is planted.

Hand planting offers more flexibility in design, but is obviously slower than planting by machine. The common tools used in hand planting are hoedads, planting bars, and planting shovels (straight, narrow blades). Lapin has used planting bars and shovels in clay soils with relative ease; the efficacy of hoedads in heavy soils is unknown. Peterson (pers. com.) reported that hoedads and bars are used in hand planting in The Big Woods Project. Either may be good tools for non-mechanical direct seeding of oak and hickory in heavy soils.

Choice of planting method(s) must consider several factors, of which efficiency of planting is just one. If weed control is to be done by mowing or band application of herbicide, trees must be planted in an arrangement that facilitates the use of machinery; that usually would indicate planting in rows. Row planting, as noted above, results in restorations that resemble plantations, which may not be desirable if an objective is to more closely simulate natural forest structure. If post-planting mechanical or band-herbicide weed control is not to be used, combinations of machine planting, hand planting, and direct seeding may be efficient. One would first need to machine-plant widely spaced rows of seedlings; direct seeding by machine could then (or before seedling planting) take place in bands; finally, hand planting could be used to lend the planting a less uniform, more natural-appearing spacing.

The way that a seedling is put into the soil is important for seedling survival and growth. In the 1920s Europeans noticed growth problems in pine plantations and investigated causes (Bergman and Haggstrom 1976). Wibeck (1923) discovered that plants with root deformations due to improper planting could show normal growth for up to 15 years, but would then deteriorate and die. Spiral root growth, or root twirl, most common in containers, can be problematic because roots may graft together and rather weak side roots are produced. Spiral roots can also “strangle” a tree after five or six years of normal or better than normal growth (Bergman and Haggstrom 1976). J-roots, a condition in which roots have been shoved into the planting hole such that the lower roots are curved upwards in a j-shape) lead to poor anchoring and mortality. Bergman and Haggstrom stated, “Propagating in containers with walls impenetrable by the roots are causes for the biggest risks for deformed root systems. The damage increases with the length of time the plant stands in the container.” They also noted that the effects of malformed root growth caused by improper planting or prolonged growth in containers are not revealed for many years. They also provided the information that roots are especially sensitive during leaf flush and plants should not be handled or transported at that time.

Tree-planting weather has been characterized as normal, marginal, or critical; no planting should be undertaken in critical conditions (Long 1991). Normal for planting southern pines is 33-75^{o}F, greater than 50% relative humidity, and less than 10mph wind; these values seem to represent a reasonable range for planting conditions in the northeast also. Critical weather in the southeast is temperature below freezing or above 85^{o}F, relative humidity less than 30% and wind speed greater than 15mph. Planting dormant seedlings in temperatures below freezing may be acceptable in the northeast, but ability to properly close holes in cold-soil conditions is a concern (Stroupe and Williams 1999). Long (1991) provided guidelines for how to treat seedlings on normal weather days, as well as on marginal days. Soil moisture below 50% field capacity is a critical condition and no planting should be undertaken if there is no significant precipitation likely in the short-term forecast.

Tree Spacing and Numbers Per Unit Area

The Big Woods Project is probably the largest effort to restore eastern deciduous forest; it appears to be the only project to date that is approaching anything of a landscape-scale active restoration in the Northeastern or Midwestern forests. One of the principal restorationists involved, forester Dick Peterson, has designed plantings of 450-600 seedlings per acre and has used 2.7 x 3m spacing for trees (Peterson pers. com.). Charlie Evenson (pers. com.), another of the long-term project restorationists, takes a different approach and plants approximately 2,200 seedlings per acre (a spacing of about 1.3 x 1.6m), but 50% of the stems he plants are shrubs. Unfortunately, very little has been written about the techniques employed in the project, and oral and electronic-mail communications have provided most of the information about Big Woods restoration.

Spacing of about 2 x 2m is traditional for plantation plantings (Godel 1982, Harmer and Kerr 1995). von Althen planted seedlings 1.5 x 3m (von Althen 1981), whereas Zaczek et al.'s (1997) plantings were spaced 1.2 x 1.2m. Typical spacing for direct-seeding southeastern oaks by machine are 0.6 x 3m to 1.1 x 4.5m (approximately 1450 seeds per acre) (Bullard et al. 1992, Schweitzer and Stanturf 1999), whereas seedling bottomland oaks have apparently been planted at approximately 3.7 x 3.7m spacing (Schweitzer and Stanturf 1999).

Smith and Strub (1991) not only provided spacing specifications for southern pines but also included useful information regarding relationships among spacing, growth, and stocking over time; the principles are certainly transferable to any species, although the numbers and graphed curves would differ. According to Smith and Strub, the decision of how many stems to plant is a function of the species, establishment costs, plantation health, the product objective, and expected survival.

Recommendations for promoting a more natural pattern which also allows for naturally dispersed propagules to establish is to plant at wider spacing (e.g. 5 x 5m (400 stems per ha)), drive mechanical planters or hand-plant in irregular patterns, and leave gaps unplanted (Allen 1997). If leaving unplanted gaps, spacing could be closer, which may lead to better growth (form?) of planted trees. Herbaceous gaps surrounded by woodland, however, could create optimal habitat for deer, and lead to inordinately high browsing in the newly created woods (Harmer and Kerr 1995). With a landscape pattern that already provides very good deer feeding and cover habitat in the Champlain Valley, the concern may not apply.

The Natural Resources Conservation Service standards have different spacing guidelines for plants of different height at maturity. The standards are: trees greater than 7.5m tall – 200-450 stems per acre (3 x 4.5m spacing); shrubs and trees 3-7.5m tall – 200-450 stems per acre (3 x 4.5 m); shrubs less than 3m tall – 450-1200 stems per acre (1.8 x 3m). See Appendix D for NRCS Conservation Practice Standards. The US Fish and Wildlife Service does not have guidelines for planting density.

Britain's Woodland Grant program seeks a minimum density of 1100 stems/ha (3 x 3m spacing), but suggests 2500 stems/ha (2 x 2m spacing) as better for creating a closed canopy and production of high-quality timber (Harmer and Kerr 1995).

Fertilization

Several restoration researchers have reported the effects of fertilizer application on seedling growth. Nutrient amounts and cycles are quite altered when forestland is converted to agricultural use, as is discussed in greater detail below. Despite the fact that nutrient amounts are usually elevated by agricultural practices, some restorationists have looked at the influence of nutrient additions to hasten seedling growth. On a very acidic site in Pennsylvania, Demchik and Sharpe (1999) found that liming had negative effects on survival and growth of red oak seedlings if it was not combined with a weed-control technique, but a positive effect if weeds were controlled. They also observed that neither phosphorous nor potassium additions appeared to be of importance to seedling growth. Similarly, in southern Indiana, on a soil that was so heavily eroded that the Wisconsinan loess no longer covered the underlying paleosol soil, fertilization with nitrogen, phosphorous, potassium, calcium, magnesium, sulfur, and calcium carbonate (lime) resulted in 47% and 60% increases in height and diameter, respectively, and nitrogen was found to be the limiting factor (Rathfon et al. 1995). Importantly, fertilizer treatment did not alter root:shoot ratios of the red oak seedlings. Of further significance, the fertilized trees were favored by browsing deer, but they did resprout vigorously. The authors noted that fertilization might delay winter dormancy, which would not be desirable. In contrast with Demchik and Sharpe’s finding of the need for weed control in association with fertilization, Rathfon et al. observed a negative interaction between fertilization and tilling; in their research, tilling reduced the ability of fertilization to induce the multiple growth flushes that contributed to the increased height growth.

Site preparation techniques can have an impact on nutrient availability and can have fertilizing effects. Mechanical site preparation can have a fertilizing effect because baring the soil allows it to warm more, and warmer soil temperatures are associated with increased nitrogen mineralization (Morris and Lowery 1988). The same authors noted that increasing nitrogen might not be important for southern pine seedlings if competition is controlled, because there is much greater mineralization than nitrogen uptake by 600-880 pines per acre. Nevertheless, the relationship of nitrogen mineralization to long-term nitrogen supply has not been quantified. Morris and Lowery therefore concluded that site preparation that leads to initial increases in growth may be transient and later growth may actually be slower, especially if mineralization potential is reduced by site preparation techniques that remove organic matter from the soil (e.g., burning). One should note, however, that Morris and Lowery's chapter is concerned with growing southern pines rapidly, not with restoring soil conditions to those more similar to a natural forest ecosystem, and they are thus concerned with fertilizing effects that can be brought about by site preparation techniques.

Potential phosphorous changes related to site preparation are probably less compared to those in the nitrogen system. Extractable phosphorous concentrations were not found to change significantly under various site preparation treatments, but burning can increase availability, and short-term gains may be of importance for seedlings with poorly developed root systems (Morris and Lowery 1988).

In a physiological study, Hechler et al. (1991) compared responses of three oak species seedlings to the interaction of drought and nitrogen fertilization. They found that bur oak (*Quercus macrocarpa*) showed an adaptation to drought if nitrogen was available, but under nitrogen-deficient conditions, the adaptation (partial stomatal closure) was not seen (full stomatal closure and thus cessation of photosynthesis resulted). Red oak also showed a tendency that indicated it needed nitrogen to conserve water, while for white oak nitrogen deficiency or addition did not alter the seedlings' response to drought. Hechler et al.'s work shows how the interactions of species traits, site conditions, and growing-season weather result in the survival and growth that is seen in the field. Unless one is interested in a few particular species, the understanding of these species difference and mechanisms would be difficult to apply in a restoration plan. Nevertheless, with such understanding, one can adapt management practices to the site and weather conditions during the critical first and second years after planting.

Soils and Forest Restoration

Land use history has a large impact on soils; farming and grazing change physical and chemical properties of soils (Harris and Hill 1995, Moffat and Buckley 1995, Compton and Boone 2000, Stanturf et al. 2001, Honnay et al. 2002). Draining, plowing, liming, fertilizing, and manuring can have marked effects on soils. Different aspects are likely to return to conditions similar to pre-agricultural norms in times ranging from decades to centuries (Harris and Hill 1995, Moffat and Buckley 1995). Among the common nutrient changes accompanying agricultural use of land are excessive levels of nitrogen, phosphorous, calcium, magnesium or boron. Physical changes due to compaction and plowing include alteration in a soil's natural porosity (can be an increase in porosity from plowing and cultivation, or a decrease from compaction), disruption of horizonation, increases in bulk density, and loss of small-scale soil structures such as tip-up mounds and pockets of deep organic layers. Changes in soil biota, including bacteria, fungi, mycorrhizal fungi, actinomycetes, protozoa, and algae) also occur, and these can be related to physical and chemical soil changes as well, for microbiota help to mediate many plant nutrient transformations and play a part in maintenance of soil structure (Harris and Hill 1995). All such soil alterations have impacts on plant growth. Soil changes may lead to habitat that is either unsuitable for certain species or life stages (e.g. seedlings) of plants or in which some species are at a competitive disadvantage. Altered physical structure can impede tree root growth (Moffat and Buckley 1995, Honnay et al. 2002).

Many vascular plant species have mycorrhizal fungal associations. In a wetland and wetland-margin flora in Connecticut, 92% of 89 species of herbs, shrubs and trees (representing 73 genera and 42 families) were found to have vesicular-arbuscular mycorrhizal fungi (Cooke and Lefor 1998). Not every species consistently had associations from site to site, and the impacts of associations or the lack of them on ecosystem function are not known. The system, however, is clearly complex and the loss of mycorrhizal fungi from a soil must have some effect on ecosystem function and plant metabolic and population processes (even for the many species that do not have obligatory need of mycorrhizal associations).

Afforestation and reforestation of previously cleared land does bring soil properties closer to those found in less altered, native forests of a landscape. Harris and Hill (1995) outline the soil change tendencies associated with woodland maturation on former agricultural or reclamation lands:

1. Litter buildup and proliferation of feeder roots in the O and A horizons.
2. Greater input of woody parts to soil and long-term immobilization of nitrogen becomes more common.
3. Internal cycling of nitrogen and phosphorous increases.
4. C:N and C:P ratios increase.
5. Illuviation of clay and mobile humus fractions increases, and therefore lowers the pH in the upper horizons.
6. Decomposer communities change, with a relative increase in fungi.

Published examples of soil changes in post-agricultural forest appear to be few, but those read do uphold the assertions of Harris and Hill. In the Lower Mississippi Alluvial Valley, five to ten years after Nuttall oak planting on Sharkey clay soils, soil carbon increased and bulk density decreased (Stanturf et al. 2001). Despite such rapid changes, Compton and Boone (2000) found that after 90-120 years of post-agricultural forest regrowth in Massachusetts, those sites still had 39% more nitrogen and 52% more phosphorous than forested soils that had never been cultivated; the secondary forest soils also retained lower C:N ratios after all those years of forest regrowth. After thirty years of birch (*Betula*)-oak forest regrowth in Britain, the decomposition subsystem was “broadly similar” to that of an ancient woodland, but the ancient woodland did have significantly greater fungal biomass and C:N ratio, and lower microbial activity.

The relative ease with which soil properties and processes can be measured has led authors to suggest that parameters such as soil carbon, microbiological activity rates, and microbiological community biomass and composition would serve well for evaluating how effectively some ecosystem functions had been restored (Harris and Hill 1995, Stanturf et al. 2001).

Herbaceous Plants and Forest Restoration

Most forest restoration projects focus on creating a tree canopy – the physiognomy of a forest. Although many restorationists may recognize the importance of restoring the herb layer, there are many fewer attempts to restore non-tree floras than tree floras. Many herb species are slow to return to intensively disturbed or highly isolated sites, and responses vary widely by species (Marks and Mohler 1985, Bratton and Meier 1998, Stover and Marks 1998, Singleton et al. 2001, Honnay et al. 2002, Vellend et al. in press). Bratton and Meier (1998) hypothesized that some plant families are more sensitive to disturbance than others, but it is unclear if the existing research upholds that hypothesis. Lists of species which appear to be better or worse at establishing in forests on abandoned agricultural land can be gleaned to some extent from the literature. Refer to the review of fragmentation effects on plants in Lapin (2003) for more detailed accounts of the available literature.

Of importance for restoration of herbaceous plants to woodlands is an understanding of what barriers a species faces (Bratton and Meier 1998). Is, for example, reestablishment of individuals the limiting factor? Or perhaps reestablishment proceeds but the plants are not able to reproduce and spread effectively in disturbed sites. Have microclimatic changes made a site inhospitable to a species? Have soil changes? Many woodland herbs may require mutual relationships, with ants or mycorrhizal fungi for example, in order to grow and spread effectively in a restored woods.

Bratton and Meier (1998) provided a list of seven types of eastern deciduous forest species they suspected were most likely to require active reintroduction or active habitat restoration. A quick perusal of the characteristics will indicate that for many, many native herbs, the information is unknown or not well documented. The listed species types are those: 1) with preferences for mesic sites, 2) dependent on gap-phase succession for reproduction, 3) with mutualists that are vulnerable to disturbance, 4) with narrow microhabitat preferences, 5) requiring the presence of organic matter patches, deep soils, high pH or well-developed microtopography, 6) with slow rates of growth or dispersal, and 7) found in very small or scattered populations.

Both Mottl (2001) and Francis and Morton (1995) conducted experiments in herb restoration. Mottl, who worked in Iowa, found that after two seasons, 12 of the 16 monitored species showed greater than 75% survival, and 9 of them showed greater than 90% survival. All were grown in pots in nurseries and then planted into the fields. Of the species with poor survival, one case was attributed to leaf spots and heavy browsing and another to rabbit or deer herbivory. Francis and Morton experimented more extensively in European woods; they sowed seed mixes at two densities and planted some pot-grown stock also. All but one species established successfully by direct seeding, and all species established successfully by planting pot-grown stock. They observed that species could be classified into two groups – one that consisted of light-demanding edge species which quickly establish and mature, and another of shade-tolerant “true” woodland species that are slower to establish and mature (on the scale of several years minimum). For the first group they recommended a moderate sowing density of 3 kg ha^{-1} (0.3 g m^{-2}); the second group apparently would require a much greater sowing density of 10 kg ha^{-1} (1 g m^{-2}). For potted plants, Francis and Morton recommended planting in single-species groups of approximately 9 individuals per m^{-2}, which more closely mimics the often seen pattern of natural clustering and helped establishment of their plants. Herbicide pre-treatement for several years was used to reduce competition. Sowing seed in fall or winter was recommended, to provide for natural stratification and vernalization.

Packham et al. (1995) worked in Great Britain woodland restorations and experimented with both seed-sowing treatments and mulch treatments. Results differed, not surprisingly, by species and by site. In general, woodland grasses were slow to start, but flowered profusely the third year, and *Geum urbanum* (an avens) was one of the most successful species introduced. Time-of-sowing experiments showed no clear advantage to autumn sowing, and the authors thus suggested that sowing at different times of the year might be advantageous for increasing chances of germination of the various species, and providing opportunity for the full range of genetic variability with respect to germination to be expressed. Packham et al. found that woodchip mulch spread over the forest floor appeared to mitigate harsh conditions (e.g., extreme moisture fluxes, lack of organic matter) in the urban-created forests in which they worked. The practicality of spreading woodchip mulch over larger areas may be prohibitive, but if herbaceous plant introduction proceeds in patches scattered throughout a restoration, it may indeed be practicable. The authors also preferred direct seeding for some herb species because of cost efficiency and the greater potential for variability and resilience, but they acknowledged that it is difficult to store seed of some species and some species did establish better from transplants.

Cornett and Evenson (1999) did some work with herbaceous species restoration in The Big Woods. Plants propagated in a nursery under a shade-house were planted on restoration sites, but success was very poor due to deer herbivory. Their nursery protocol, however, of collecting wild plants and then growing them in the shade-house as “mother plants” to produce seed or for vegetative reproduction, was successful.

Another technique for restoring herbs is transplanting soil. Van der Valk and Pederson (1989) found that donor soils could be used to rapidly establish a species-rich vegetation dominated by native species; in their experiment, the removal and immediate placement of 5 cm of topsoil gave the best results on a mined site in Western Australia. Wilhelm et al. (1987, cited in Van der Valk and Pederson 1989) observed that on the Des Plaines River bottomland in Illinois best results were obtained when soil was collected at the beginning of the growing season and to a depth no greater than 25 cm.

Van der Valk and Pederson listed several questions one would want to know in order to implement the donor soil technique effectively: What time of year should soils be collected? What length of time can soil be stockpiled without substantial loss of seed viability? How deep a layer is needed? What environmental conditions must be established and maintained to ensure adequate recruitment? What can be done to prevent the establishment of unwanted species? In addition, they noted that watering experiments showed that soil moisture had a major effect on species germination, and that fertilizing, liming, or incorporation of organic matter could be used to alter site conditions to foster the recruitment of desirable species. Johnson and Bradshaw

(1979) and Lyle's (1987) surface mine reclamation manual were cited as useful references regarding donor soil and seed bank matters.

An understanding of temperate deciduous forest seed banks is indispensable when considering whether and how to employ donor soil transplants. Pickett and McDonnell's (1989) chapter in "Ecology of Soil Seed Banks" is an excellent reference with which to begin. Their summary implied that donor soil transplantation in upland forests would be unlikely to result in establishment of forest herbs, for forest herbs are rare or completely lacking in eastern deciduous forest seed banks. The species most common in seed banks are ones that characteristically establish on open sites, and in the lists and data tables provided in the chapter, the genera *Prunus, Betula, Rubus, Carex,* and *Juncus* seemed to recur. The seed banks and the standing vegetation usually differ markedly in eastern deciduous forest and field vegetation; only in repeatedly disturbed arable fields have seed bank and standing vegetation composition sometimes coincided.

Although upland soil transfers appear unlikely to be a good approach to herbaceous plant restoration, actual transplantation of forest floors may be effective. Of course, that should only be attempted if the donor site were scheduled to be cleared. No papers on that technique were read for this review, however.

Donor soil transplants may be more successful for wetland ecosystems. Van der Valk and Pederson (1989) included some information on donor soil for wetlands; that paper and Leck's (1989) "Wetland Seed Banks" chapter in "Ecology of Soil Seed Banks" should be consulted for an introduction to that topic.

Animals and Forest Restoration

Most ecosystem restorations are primarily, or perhaps preliminarily, concerned with restoring native vegetation to a site. The restoration of animal species, it is presumed and hoped, will follow naturally, or perhaps with some human assistance, once the vegetation proceeds to a state more similar to the native condition. In forest restorations, the growth of a tree or tree and shrub canopy is a first step that contributes to the creation of appropriate microclimates and physical structures necessary for many forest animal species.

Forest and shrubland birds appear to principally require appropriate vegetation structure in order to select sites for feeding, roosting, and especially breeding. The literature regarding that is rather extensive and has not been reviewed here. The Vermont Institute of Natural Science (VINS) and the Forest Service Northeastern Experiment Station, Amherst, Massachusetts, are excellent resources for pertinent avian information. Some restorationists in the Lower Mississippi Alluvial Valley have recommended the use of fast-growing, early successional species to promote rapid colonization of migrant birds (Twedt and Portwood 1997). Twedt and Portwood found that bird species diversity was increased fourfold (36 vs. 9 species in plantings 4-7 years) at sites where cottonwood (*Populus deltoids*) was planted along with oaks, as opposed to sites with only oaks planted. Another fast-growing species of relevance to the Champlain Valley that they mentioned was green ash, but one can also consider that birches, willows (*Salix* spp.) (tree and shrub species) and elm (*Ulmus americana*) are appropriate for clayplain and floodplain forest restorations.

Flowerdew and Trout (1995) summarized expected population trends of small mammals in restored woods. They noted that starting conditions (e.g., recent land use) is likely to affect the small mammal community to a great extent. On fertile agricultural lands, species-rich communities and high-density populations would be expected in the luxuriant herbaceous vegetation. In fields where a mowing regime prevents the development of a diverse and complex herbaceous layer, small mammal communities are expected to be rather depauperate. The masting characteristics of the tree species present will likely be dominant factor affecting rodent density, and as a new species composition and vegetation structure develop at a restoration site, the rodent species may be affected by changes in the herb layer and in (hard and soft) masting species more than the insectivore species. Studies on rodents in highly fragmented agricultural landscapes with and without “good” connectivity have much basic information to contribute that is pertinent to thinking about small mammal responses to restoration; see Chapter Two of Lapin (2003) for a review of that literature.

Strategies for salamander (and herb) restoration were presented in Bratton and Meier (1998). They recommended selecting mesic sites next to streams, establishing deeper pockets of litter, changing soil chemistry if necessary, and introducing mutualists (the latter one or two are perhaps more important for herb establishment and spread). The importance of coarse woody debris as a component of salamander and invertebrate habitat has been well documented (as a starting point see Maser et al. (1979), McComb and Lindenmayer (1999) and Andrews (2002), and search for articles authored by P. deMaynadier or K. Pollock for further information and citations). Therefore, it seems reasonable to suggest that implementing techniques to create greater amounts and hasten development of coarse woody debris would be of use for salamander restoration.

With regard to invertebrates, Key (1995) stated that the likelihood of invertebrate introduction into new woods was low because the ecology of most species is poorly known, but his chapter is full of information about woodland-invertebrate relationships. Key noted that old woodland is the richest habitat for terrestrial invertebrates and that particular assemblages are associated with different successional seres. Comparison of British parks established in the 17th and 18th centuries with those that were established in medieval times showed that although there were many old trees in the former, those parks still did not appear to have the rich invertebrate fauna of the more ancient woods; it is unknown how much his analysis controlled for different ecosystem types, landscape configurations, or land-use histories. A limiting factor for many invertebrates appears to be slow colonization into new woods; some invertebrates, like mollusks are limited in mobility, while others are apparently reluctant to move. The “commoner, less fastidious species” are the ones that are highly mobile and will rapidly colonize new plantings as soon as appropriate conditions develop (Key 1995). Key provided nested lists of microhabitat features that are important to woodland invertebrates; they are listed within seven major habitat features: trees and shrubs, herbs, bryophytes, fungi, litter, soil, and water. He also presented data showing that certain tree species support substantially higher invertebrate species diversity than others. For foliage and sap-sucking species, willows, oaks, birch, hawthorn (*Crataegus* spp.), aspen (*Populus tremula*.), and elm *(Ulmus* spp.) are higher in diversity; beech (*Fagus sylvatica*), linden (*Tilia* spp.), ash, field maple (*Acer campestre*) and hornbeam (*Carpinus betulus*) are lower. For wood-eating species, oaks, beech, elm, birch, willows and linden are especially important. Such information highlights the importance of planning for inclusion of many plant species and many habitat features in a restoration.

One specific study of invertebrate response to restoration is Marzelli's (1995) work in Europe. The large marsh grasshopper was the species of interest, and Marzelli found that a factor limiting recolonization was the presence or absence of areas with the appropriate soil moisture content for development of eggs. Restored areas with high population densities were characterized by a mosaic of bare ground, sparse vegetation, and dense vegetation; the mosaic was seen to be important for providing a variety of microclimates conducive to all stages of the grasshopper development. The message of the study reiterates much of what Key presented, that a diversity of microhabitat structures is very important, even for a relatively highly mobile invertebrate, to allow for reproduction and persistence in a restored ecosystem. Complex life histories, needs for different habitat conditions during different life stages, and our ignorance of most species' requirements are likely to be themes heard repeatedly with respect to invertebrate restoration.

Squirrels are important in the dispersal of acorns, hickory nuts, and other nuts, and an interaction of vegetation physiognomy and small mammal behavior is of interest to forest restoration in the Champlain Valley. Gray squirrels (*Sciurus carolinensis*) and mice were seen to cache many times more acorns in a mixed oak stand than in an adjacent red pine (*Pinus resinosa*) stand (Thorn and Tzilkowski 1991). Wessels (pers. com.) has observed that squirrels more frequently cache acorns on mounds than on other parts of the forest floor. The behavior of seed dispersers in relation to vegetation cover, landscape pattern and microtopography is thus clearly of importance, and planning for restoration in old-field pine stands and conifer plantations should consider information such as this that indicates nearness of seed source may not be all that is required for restoring animal-dispersed species to such areas.

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5. ECOSYSTEM STRUCTURE AND PROCESSES AND FOREST RESTORATION

Exotic Species Invasion and Forest Restoration

There is an abundance of literature on invasive species and it was beyond the scope of this project review it thoroughly. Also, there was no attempt to do a literature search focused specifically on the ecological impact or effective control measures of the most problematic exotic species in the Southern Lake Champlain Valley. Instead, this section is an overview of ecological considerations of invasive species important to consider when conducting ecological restoration.

Invasive species can influence ecosystem structure and function in a number of ways including: 1) changing composition and physiognomy, 2) displacing native species, 3) altering soil processes and nutrient cycling, 4) changing the disturbance regime, and 5) altering hydrology (Vitousek 1990, Kenna et al. 1999). Identifying which species have the greatest potential to influence structure and function is important for establishing priorities for controlling invasives in ecological restoration projects. Farnsworth and Ellis (2001) recommended measuring multiple parameters to effectively characterize the impact of an invasive species. They found clear evidence, despite contradictory reports from the literature, that purple loosestrife (*Lythrum salicaria*) had a negative impact on the total abundance of surrounding species. In this case, the authors recommended that standing biomass was an effective indicator of purple loosestrife’s potential to outcompete other plant species.

Understanding how a species impacts an ecosystem often does not directly lead to effective control efforts, however. Frequently species reinvade after they are controlled (Berger 1993), or they can be replaced by another invasive species. Obviously, the appropriate control technique(s) must be based on the attributes of the ecosystem that is invaded and the characteristics of the invasive species. Understanding the mechanisms leading to a species’ abundance is often necessary for determining effective means of controlling an invasive species, but frequently there is no attempt to do this (Hobbs and Norton 1996).

A variety of techniques have been recommended to make a site less suitable for an invasive species, including fire, shading, altering soil conditions, or substrate removal (Berger 1993). A recent study (presented anecdotally) demonstrated how understanding the factors that lead to an invasive species success could help to design restoration measures to effectively control the species. In the upper Midwest, the successful restoration of sedge meadows in prairie potholes has been thwarted by the dominance of reed canary-grass (*Phalaris arundinacea*), which outcompetes native sedges. Repeated applications of herbicide were shown to eliminate reed canary-grass, but prevented sedge establishment as well. In a series of experiments, Perry and Galatowitsch (2002) evaluated whether reduced light levels or nitrogen levels would result in enhanced success of porcupine sedge (*Carex hystericina*). Their results showed that lowering nitrogen through the addition of carbon (in this case sawdust) resulted in increased biomass of the porcupine sedge relative to reed canary-grass. Further experimentation is necessary to determine how carbon enrichment and vegetation harvest treatments might effectively lower nitrogen and limit the success of reed canary-grass in the field and contribute to the long-term restoration of sedge meadows on a large scale (Perry and Galatowitsch 2002).

Efforts to understand and curb an invasive species' establishment or spread can be the most effective means of addressing exotic species problems (Kenna et al. 1999). Thus, early detection and prompt countermeasures are advised (Berger 1993).

In clayplain forests of Vermont three non-native shrubs are considered to be the most problematic exotic species. They include: Common *buckthorn (Rhamnus cathartica*), European bush honeysuckle (*Lonicera morrowii*), and Tartarian honeysuckle (*L. tartarica*) (Lapin 2003). In floodplain forests in Vermont, goutweed (*Aegopodium podagraria*), garlic mustard (*Alliaria petiolata*), ground-ivy (*Glechoma hederacea*), dame's rocket (*Hesperis matronalis*), moneywort (*Lysimachia nummularia*), Japanese knotweed (*Polygonum cuspidatum*), common buckthorn and the two aforementioned honeysuckle shrubs are considered the most problematic exotic plant species (Thompson and Sorenson 2000). Based on limited data from floodplain plots in the Lower Poultney and Hubbarton Rivers (Sorensen et al. 1998), moneywort was the only invasive, exotic species with coverage greater than 0.5%. It reached 5% in the Hubbarton River plot and had a frequency of 75% (present in 3 of 4 plots). Sorenson et al. did not note major invasive species problems in the four floodplain forests they sampled; they did, however, indicate that there was potential for invasives to become more serious problems, based on observations in other floodplain forests in Vermont. It should be noted that they sampled only the highest quality sites in the area; thus, the plots were a nonrandom sample of floodplain sites, and recently disturbed sites (a number of which have been observed at recently acquired TNC lands on the Lower Poultney River) are under-represented. Several of the most problematic invasive species (including goutweed and Japanese knotweed) reach considerable densities in the Upper Poultney River floodplains. Early detection and efforts to identify problems before they spread (such as those being developed and undertaken by the SLCV office of TNC (Droege pers. comm.)) are warranted. Such efforts are especially critical because riparian habitat is highly susceptible to invasion, and waterways can serve as an effective means of spreading propagules (Berger 1993).

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6. STRUCTURE AND DYNAMICS OF CHAMPLAIN VALLEY CLAYPLAIN AND FLOODPLAIN FORESTS

Information Specific to Valley Clayplain Forest
(Material in the following sections is excerpted or paraphrased from Lapin (2003).)

Species Composition

Overstory species composition was derived from a sample of 28-0.1ha sample plots in the southern Champlain Valley (Table 1). Data are from woods that were never cleared for agriculture, but were disturbed during a two-century period of use as farm woodlots and woodland pastures. Compositional differences can be seen between the clay-soil forest type and the sand-over-clay soil forest type. Differences between the mesic and wet-mesic clay-soil types are also apparent. In planning species mixes and percentages for a restoration site, mapping out areas with different soil moisture regimes and the presence or absence of a sandy overburden is recommended.

Composition of the contemporary forest fragments may be substantially different from that of the presettlement forest (Table 2). The most striking differences are the paucity of beech and the preponderance of red maple (*Acer rubrum*) in today's forests. It is likely that red maple was historically common in the wet-mesic and sand-over-clay clayplain areas, but sugar maple was likely to have been the more abundant maple in mesic clayplain forest. Since initial surveyors did not typically note which maple (or at least that information is not often transferred into the published historical ecology literature) this is only suspected, it is not known for certain. Hemlock percentages are also somewhat lower in the modern forests. In addition to the increase in maples, oaks and ash show a slight increase also.

Table 1. Relative overstory density of clayplain forest types. Species arranged in descending order of all clay-soil plots.

	Forest Type			
Species	Mesic Clay	Wet-Mesic Clay	All Clay-Soil Plots	Sand-Over-Clay
Acer ruburm	0.234	0.352	0.302	0.276
Tsuga canadensis	0.160	0.097	0.147	0.414
Pinus strobus	0.114	0.085	0.097	0.005
Carya ovata	0.019	0.105	0.054	0.000
Fagus grandifolia	0.074	0.022	0.046	0.064
A. saccharum	0.081	0.009	0.043	0.000
Fraxinus americana	0.059	0.033	0.042	0.034
Tilia americana	0.026	0.058	0.034	0.000
Quercus rubra	0.051	0.023	0.034	0.039
Q. alba	0.041	0.017	0.033	0.005
Q. bicolor	0.018	0.049	0.033	0.000
Ostrya virginiana	0.036	0.023	0.033	0.000
Fraxinus nigra	0.014	0.056	0.028	0.000
Populus tremuloides	0.024	0.001	0.012	0.000
Ulmus americana	0.014	0.012	0.011	0.000
Populus grandifolia	0.006	0.017	0.010	0.054
Betula papyrifera	0.005	0.001	0.008	0.005
B. populifolia	0.000	0.001	0.008	0.000
Rhamnus cathartica	0.009	0.009	0.007	0.000
Q. macrocarpa	0.000	0.016	0.007	0.000
B. lenta	0.009	0.003	0.005	0.049
Carpinus caroliniana	0.000	0.006	0.002	0.000
B. alleghaniensis	0.004	0.000	0.002	0.054
Fraxinus pennsylvanica	0.000	0.003	0.001	0.000
A. platanoides	0.001	0.000	0.001	0.000
Carya cordiformis	0.000	0.001	0.001	0.000

It would be difficult to plan restoration-planting composition on the presettlement data, because beech has very low survivorship in open field plantings (Derleth pers. com.). Considerations for “balancing” the available presettlement composition data and the modern data should include: 1) reducing contemporary red maple numbers in mesic (not wet-mesic) areas, while increasing sugar maple, 2) increasing contemporary American elm densities, and 3) increasing contemporary hemlock densities.

Table 2. Clay-soil presettlement forest composition and species ranks from Chittenden County (Siccama 1971) and Town of Middlebury (Perine 1975).

	Chittenden County 1800		Middlebury 1790	
Large Trees	% Composition	Rank	% Composition	Rank
Hemlock	12.4	3	27.4	1
Beech	40.9	1	20.2	2
Maple	19.1	2	19	3
Pine	7.8	4	4.8	4
Other	0		4.8	-
Oak	1.6	9	3.5	5
Basswood	4.6	6	2.4	6
Ash	2.4	7	2.4	6
Birch	5.4	5	1.2	8
Elm	2.4	7	*	-
Spruce	0.8	10	*	-
Aspen	0.3	11	*	-
Small Trees				
Hophornbeam	2.4	-	3.5	-

* species not mentioned and presumably lumped into "other"

There is much interest in the oaks of the clayplain forest, and it is therefore worth speculating about differences in the contemporary and presettlement datasets. Siccama's presettlement oak abundance (1.6%) was substantially lower than Perine's figure (3.5%). Lapin's data indicated a two- to four-fold increase in oak abundance. Cogbill et al.'s (2002) abundance surface maps show from 5-20% oak in the Champlain Valley clay plain; Siccama's and Perine's data would thus appear to be on the low end of oak abundance on the clay plain. Oaks may in fact not have increased much at all over the clayplain landscape since presettlement forest times. Regarding the oaks, it is especially noteworthy that presettlement data are relative density data. Data from Lapin's sampling consistently ranked oaks substantially higher in relative dominance (i.e., basal area) than in relative density; thus, there may not have been many oak trees in the forest, but those present were large. This must be kept in mind when reckoning the observation of an early surveyor that Charlotte had an abundance of white oak, with the apparently contradictory witness tree data (Siccama 1971). To further speculate, it is worth considering what ecological role, perhaps that of keystone species, the massive, fecund oaks historically played in the clayplain forest. Given the lack of other guidelines and the potential “keystone-like” function of large, heavily masting oaks, it is recommended that their contemporary densities be used to guide composition objectives, rather than the lower historical densities. Future analyses by Cogbill of surveyors’ data for “clayplain towns” may suggest a different strategy.

Ecosystem Disturbance Dynamics

In contemporary thinking about restoration, the objective of restoring a natural disturbance regime is of paramount importance (Jackson et al. 1995, Hobbs and Norton 1996, Holmes and Richardson 1999). Furthermore, restoration can help contribute to adequate function of landscape-scale ecosystem dynamics and disturbance regimes (Noss and Cooperider 1994, Bullock and Webb 1995, Hobbs and Norton 1996, Noss 1996, Spies and Turner 1999).

No historical ecology or natural disturbance regime research has been published for the clayplain forest natural community, but from anecdotal evidence and extrapolation from research in other parts of the northern deciduous forest region it is possible to postulate the characteristics of the native disturbance regime. Based on these lines of evidence, it seems that the primary natural forest disturbance in the clay plain has been small-scale blowdown and ice damage. Direct evidence for the role and pattern of ice damage comes from the January 1998 ice storm that stretched from New York to Maine, including southern Quebec and the Maritime Provinces. That storm had a very patchy nature, and its particular pattern in the southern Champlain Valley was one of moderate to severe damage on higher parts of the landscape, including higher ridges of the valley’s undulations, and light or no damage in the lower parts of the landscape. In the much less undulating northern Champlain Valley of Quebec, there was widespread damage to forest fragments that was unlike the topographically related pattern in the southern part of the valley. Hence, the January 1998 ice storm was a large regional storm with a very extensive total footprint that brought differential levels of tree damage to different parts of the Champlain Valley.

Lapin has observed evidence from a similar pattern of patchy damage in the Champlain Valley and lower elevations of the western Green Mountains from a 1950 “hurricane” or downburst that came through western Vermont. Additionally, the remnant of Hurricane Floyd that arrived in the Champlain Valley in late September-early October 1999 brought limited damage to clayplain forest fragments, on the scale of single- or several-tree blowdowns.

In The Nature Conservancy’s Williams Woods, a parcel containing some of the oldest clayplain forest known, a hurricane damage report indicated that, during Hurricane Floyd, the eastern part of the woods suffered substantially more damage than the western portion; the younger woods in the northern area had little blowdown or crown damage (Ruesink 1999). A wind research analyst entered the following in the William Woods’ comments book:

> I’m a wind research analyst with primary work centering on hurricanes. These woods show signs of the September 1938 hurricane (southeast winds), the November 1950 hurricane (east and northeast winds), Hurricane Hazel in October 1954 (south and southwest winds), Hurricane Aubrey in June 1957 (south and southwest winds), Hurricane Frederick in September 1979 (south, southwest, and west winds), the January 1998 ice storm (top and limb damage), and prolific Hurricane Floyd damage from September 1999 (north and northwest winds) (Valentiejn 1999).

It is interesting that Valentiejn assessed Floyd’s damage as “prolific,” yet according to her casual observations the small patch of woods also showed previous damage from five other strong wind events in the same century. It may be that fresh storm damage looks to be more “prolific” than damage from older storms. For an accurate picture of damage, it is necessary to document what remains standing as well as what has blown down. There were obviously plenty of large trees that remained after each of the previous storms, and post-storm plot sampling during the present research showed that a closed canopy with small gaps remained as the post-Floyd forest structure. Hence, casual reports must be interpreted with some skepticism.

In summary, the anecdotal evidence suggests a disturbance regime with the following characteristics: predominantly wind- and less frequently ice-storms, a very patchy footprint, frequent light to moderate damage, and no indication that the disturbances of the past century have been of an intensity to be “stand-regenerating” events.

Works of Cogbill (2000), Keddy (1994), Marks and Gardescu (1992), Seischab (Seischab 1990, Seischab and Orwig 1991, Seischab 1992), and Whitney (1982) indicated that stand regenerating disturbances in northern New England, Ontario, the Adirondack and western New York regions, and Ohio were limited to small patches and were infrequent. Seischab and Orwig (1991) postulated that in the western New York till and lake plains the gentle terrain was not likely to generate the air turbulence needed for large-scale windthrows. Based on original survey data for that region, both Marks and Gardescu and Seischab reported that presettlement windthrow occurred on steep slopes on the Alleghany Plateau, and not in the lowlands. Whether Seischab and Oriwg’s assessment was altered by the occurrence of extensive wind damage in the Five Ponds Wilderness area in the flats of the western Adirondack Mountain region, which occurred several years after their 1991 publication, is unknown. The landscape position of Five Ponds, a rather flat area adjacent to mountains, is more similar to that of the Champlain Valley. The Five Ponds storm clearly indicates that there is at least the potential for very extensive damage on the scale of 1,000s of hectares in the New York-Ontario-Vermont-Quebec region.

Natural fire is unlikely to have been a major disturbance in the clay plain. There is, however, evidence that historic fires occurred in hemlock-pine forests on steep slopes of the Green Mountain western escarpment (Mann et al. 1994). Also, it is likely that fire is natural on the exposed western shores of Lake Champlain (since western lake shores are more prone to lightning strikes (Barnes pers. com.) and the dry, often shallow soils on the lake bluff support vegetation that is more flammable). Nevertheless, it is unlikely that fire was an important process in the clayplain forest. Cogbill (2000) noted that although the Hudson-Champlain corridor was the area where fire was historically most frequent in northern New England and New York, catastrophic fires were evidently restricted to sandy or rocky substrates. It is hard to imagine that dry, flammable fuel loads were adequate in the clayplain forest to sustain fires that could have spread from hot burns on adjacent lands such as the shores of Lake Champlain, the dry, steep, western mountain slopes, or the Chittenden County sand plains. Furthermore, there is no indication that Native Americans of northern New England (Whitney 1994) or southern Ontario (Campbell and Campbell 1994) significantly used fire as a widespread forest management tool, although they may have practiced burning to manage their agricultural fields (Campbell and Campbell 1994).

Pehr Kalm ([1770] 1937), one of the first botanically oriented Europeans to travel through the Champlain Valley, wrote that the lowland was extensively forested, but the coniferous mountain forests were burned in places. On page 569, his diary entry mentioned that the natives are careful to put out their fires, especially in the summer, so as to avoid burning the forest, but on pages 374 and 388 he noted that the mountain forests have in some places been destroyed by fire. The country on the east side of Lake Champlain was described as “low and flat and covered with woods” (p. 388). Regarding the coniferous mountain forests observed during his stay at Crown Point (on the Champlain Valley lake plain in New York), Kalm wrote, “One of the chief reasons for their decrease is the numerous fires which happen every year in the woods, through the carelessness of the Indians, who frequently make great fires when they are hunting, which spread over the fir woods when everything is dry” (p. 374). Kalm’s anecdotal evidence suggests that fires were more frequent in the coniferous forests, where fuel loads are higher, and perhaps negligible in the deciduous forests on the valley floor, but the use of fire as a management tool by Native Americans cannot be fully dismissed.

To summarize, the natural disturbance regime of Champlain Valley clayplain forest appears to be one of small- to moderate-sized, ice- and wind-mediated events of varying intensities. Fire does not appear to have played a major role in the clayplain forest. The inferred assessment indicates that there is no need to replicate any disturbance processes at the onset of forest restoration. As a forest canopy develops, it may become advantageous to artificially create a variety of dead-wood structures, small gaps, and tip-up mound microtopography. Since establishing forest canopy is a primary concern, it should probably be at least 40 years of tree growth before any gaps are artificially created in a restoration started from open field. A heterogeneous microtopography and dead-wood structures can be designed into a restoration at the start, and can also be created later by mechanically "forcing" tip-ups and by girdling and felling trees.

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Information Specific to Floodplain Forests of the Champlain Valley

Restoring floodplain forests are a priority in the southern Champlain Valley because extensive portions of the once continuous floodplain forest were cleared for agriculture over 200 years ago and because a riparian buffer is essential for maintaining and improving water quality. The Lower Poultney River is home to a diversity of rare and endangered aquatic species that depend on clean water for their survival. Much of the land that would support floodplain forest remains prized for agriculture due to high soil fertility, good moisture retention, and the absence of rocks. Therefore, unlike much of the forestland in Vermont, which has over time been converted back to forest, a relatively low percentage of the area that would in its natural state be floodplain forest is forested today (Thompson and Sorenson 2000).

Three riverine floodplain forest types and their successional variants and one lakeside floodplain forest type have been described in the southern Lake Champlain Valley region (Thompson and Sorenson 2000). In the Vermont natural community classification, the floodplain forest natural community types are named as follows: Lakeside Floodplain Forest, Silver Maple-Ostrich Fern Riverine Floodplain Forest, Silver Maple-Sensitive Fern Riverine Floodplain Forest and Sugar Maple-Ostrich Fern Riverine Floodplain Forest. The three riverine floodplain forest types correspond closely to types proposed for the New England Regional floodplain forest classification (Sperduto et al. 2002). In the New York natural community classification, there is a single broadly defined floodplain forest type that encompasses considerable variation (Edinger et al. 2002).

Silver Maple-Ostrich Fern Riverine Floodplain Forest

Silver Maple-Ostrich Fern Riverine Floodplain Forest occupies active floodplains of moderate gradient portions of the main stem of major rivers. In New England this type is restricted to the western part of the region, only extending as far east as the Connecticut River valley (Sperduto et al. 2002); it also occurs in New York, Pennsylvania, Maryland, Quebec, and Ontario with similar forests in Midwestern states (Thompson and Sorenson 2000).

Forests of this type are typically considered a wetland and generally receive annual overbank flooding, although water tables may be well below the ground surface for most of the growing season (Thompson and Sorenson 2000). Common micro-topographic features include levees and old meander scars. Soils are alluvial and in Vermont this forest type has typically been observed on well to moderately-well drained sandy loams that lack mottles in the upper horizons. A surface organic layer is commonly absent because of annual mineral-soil deposition.

Large trees and a robust herbaceous understory are indicative of the favorable growing conditions on the rich alluvial soils. The vegetation of this forest type is characterized by tall (up to 30 m) silver maples and a near absence of shrubs and subcanopy trees (Thompson and Sorenson 2000, Sperduto et al. 2002). Cottonwood may be co-dominant on some sites and American elm, slippery elm (*Ulmus rubra*), hackberry (*Celtis* occidentalis) may also be present. Black willow (*Salix nigra*), sycamore (*Platanus occidentalis*) and box-elder are frequently present in early- to mid-successional forests; butternut (*Juglans* cinerea) occasionally is dominant on abandoned agricultural land (Thompson and Sorenson 2000).

The herbaceous layer is often luxuriant and tall with ostrich fern (*Matteuccia struthiopteris*) strongly dominating the ground-cover with wood nettle (*Laportea canadensis*) interspersed Thompson and Sorenson 2000). Sperduto et al. (2002) recognized two variants, one dominated by ostrich fern and the other dominated by wood nettle. In the ostrich fern variant, ostrich fern is more abundant than wood nettle and box-elder, white snakeroot (*Eupatorium rugosum*) and jewelweed (*Impatiens* spp.) are common. In the wood nettle variant, wood nettle is more dominant or is equally abundant as ostrich fern and poison ivy (*Toxicodendron radicans*). Whitegrass (*Leersia virginica*), false-nettle (*Boehmeria cylindrica*), and stout wood-reed (*Cinna arundinacea*) are more frequent or abundant than in the ostrich fern variant.

In the southern Lake Champlain Valley, quantitative data have been collected by Jim Graves from forests classified as this type (or a successional variant) on the Green Mountain College campus and on private land in Fair Haven (Field et al. 2001). The dominant species on the Green Mountain College site are cottonwood and sycamore with an understory dominated by *Impatiens* spp. and ostrich fern. At the Fair Haven site, dominant species are box-elder and silver maple. Ostrich fern, and to a much lesser extent *Impatiens*, are abundant in the ground-cover.

Silver Maple-Sensitive Fern Riverine Floodplain Forest

Silver Maple-Sensitive Fern Riverine Floodplain Forest occupies floodplains of lower gradient portions of larger rivers, as well as depressions in floodplains of more moderate gradient reaches. This type is found throughout New England (Sperduto et al. 2002) and extends into New York, Pennsylvania, Quebec, and Ontario (Thompson and Sorenson 2000).

These forests are always considered to be wetland and are annually flooded, some years in both spring and fall. The alluvial soils (in at least the Vermont occurrences) are commonly fine in texture; they are often silt loams, or less frequently clay loams or very fine sandy loams (Thompson and Sorenson 2000). Mottling in the upper four inches of soil is typical, and a surface organic layer is lacking. It has been observed in Vermont that the Silver Maple-Sensitive Fern Floodplain Forest type generally occurs on siltier and wetter soils compared with the Silver Maple-Ostrich Fern type; however, Sperduto et al. (2002) note that this pattern is not so evident in New Hampshire occurrences.

Silver Maple is dominant with green ash, a co-dominant especially in the Champlain Valley of Vermont (Thompson and Sorenson 2000). American elm and less commonly swamp white oak (*Quercus bicolor*) may co-occur, and black willow is a common successional species in this forest type. Shrubs are typically sparse, although winterberry holly (*Ilex verticillata*) may occasionally be abundant in patches. There is commonly more silver maple in the subcanopy than in the Silver Maple-Ostrich Fern Floodplain Forest type (Thompson and Sorenson 2000, Sperduto et al. 2002). Sensitive fern frequently forms an extensive ground-cover, with other species often being infrequent. Thompson and Sorenson (2000) reported that at most sites false-nettle is characteristic, but seldom abundant, while wood nettle may be abundant in patches in some sites and absent from others. Sperduto et al. (2002) observed that while sensitive fern and false-nettle are “common and usually abundant… there is considerable variation in abundance of the two primary herbs” and ostrich fern and wood nettle are both rare.

In the Southern Lake Champlain region, quantitative data have been collected from this forest type along the lower Hubbarton River and along East Bay in West Haven, VT (Sorenson et al. 1998, Field et al. 2001). Green ash and occasional box-elder and slippery or American elm dominated sites along the lower Hubbarton River (Sorenson et al. 1998, Field et al. 2001). The most common understory species were wood nettle, sensitive fern, riverbank wild-rye (*Elymus riparius*) and giant goldenrod (*Solidago gigantea*). East Bay floodplain forest sites on a levee and a lower floodplain site were described. Green ash, American elm and basswood dominated the levee forest. Patches of tall shrubs, nannyberry (*Viburnum lentago*), box-elder and arrow-wood (*V. recognitum*) formed a mosaic with the tall herbs, wood nettle, giant goldenrod, sensitive fern, and at least 30 species of less abundant forbs, ferns, and sedges. Vines, including Virginia creeper (*Parthenocissus quinquefolia*) and riverbank grape (*Vitis riparia*), were abundant. The lower floodplain forest at East Bay was dominated by a closed canopy of silver maple and green ash with a dense layer of sensitive fern and some wood nettle. The levee soils were coarser in texture, and both mottles and the water table were farther from the surface (Sorenson et al. 1998). Information describing forests that are presumably of this type has been collected at the Putnam Creek Marshes, NY and Jones Point, Willsboro, NY (NY Element Occurrence Data provided by G. Edinger). All three sites were dominated by silver maple and green ash was listed as an associate at two sites. Other co-occurring trees included red maple, bitternut hickory, swamp white oak, basswood, sugar maple, American elm, hackberry, black ash (*Fraxinus nigra*) and black willow, cottonwood and butternut. Sensitive fern was listed as a common understory species at each site, but the associated woody and herbaceous understory species varied considerably.

Sugar Maple-Ostrich Fern Riverine Floodplain Forest

The Sugar Maple-Ostrich Fern Riverine Floodplain Forest type is associated with higher-gradient rivers and perhaps drier topographic positions such as terraces above lower gradient rivers; extant occurrences of this type on terraces are nearly completely converted to agriculture (Thompson and Sorenson 2000). The type occurs throughout New England, New York, and New Jersey, and possibly in Quebec and Ontario (Thompson and Sorenson 2000). Sperduto et al. (2002) recognized two variants of what they called Sugar Maple(-Sliver Maple)-Ash-Ostrich Fern Floodplain Forest, one of which appears to be restricted to Vermont.

This forest type occurs in topographic situations that are less frequently flooded than are those occupied by the other floodplain forest types. Soils are typically categorized as well drained to moderately well drained, fine sandy loam over sandy subsoil, and a surface organic layer and distinct horizons are generally lacking. The type has been observed primarily on calcareous substrate in Vermont (Thompson and Sorenson 2000). The majority of examples of this type are not considered to be wetlands.

Canopy species typical of Sugar Maple-Ostrich Fern Floodplain Forest in Vermont include sugar maple, white ash, and basswood, but a number of other species may be present including butternut, American elm, musclewood (*Carpinus caroliniana*), black cherry (*Prunus serotina*), box-elder, sycamore and others (Thompson and Sorenson 2000, Sperduto et al. 2002). In the variant of this community type that occurs in Maine and New Hampshire, silver maple is often co-dominant with sugar maple and white ash (Sperduto et al. 2002). This forest type typically has more structural diversity than the aforementioned silver maple floodplain forest types; with a much higher stem density in the subcanopy and shrub layer, yet still shrub cover is low overall (Thompson and Sorenson 2000). Vines are common including Virginia creeper and poison ivy. Ostrich fern is a dominant in the species-rich herb layer.

Data from this forest type has been collected in the lower Champlain Valley from the upper Hubbarton and Poultney Rivers by Jim Graves (Field et al. 2001). Sugar maple and ash were co-dominant at each site with a variety of other canopy associates including cottonwood, basswood, musclewood and other species. Ostrich fern was a dominant with a diversity of other herbaceous species.

Lakeside Floodplain Forest

Lakeside Floodplain Forest has been described as occurring along Lake Champlain and occasionally on the shores of Lake Memphramagog. This forest type occurs in lake coves within a mosaic of wetland forest and marsh. Sperduto et al. (2002) did not recognize this type and its regional distribution is unclear (Thompson and Sorenson 2000). Lake-floodplain hydrologic regimes differ from those in riverine floodplains, and one would presume that similar forests occur on the lake margins of the Great Lakes. The National Vegetation Classification (Sneddon et al. 1998), however, did not segregate a lakeside type either.

Lakeside Floodplain Forest experiences longer periods of inundation than riverine floodplain forests, and the fine-textured soils – silt or clay loams – may hold water throughout the growing season, even after lake levels have dropped (Thompson and Sorenson 2000). Saplings and seedlings often suffer high mortality because of extended periods of inundation into the growing season. Unlike riverine floodplain forest soils, which typically lack a surface organic layer, surface organic layers may be several inches thick in the lakeside forests.

Silver maple and green ash are canopy dominants, while swamp white oak, cottonwood and American elm are frequent associates. There is considerable heterogeneity in the herb, shrub and sapling layers, and in the relative abundance of trees, due to slight elevation differences and associated duration of inundation. Sensitive fern is typically an abundant species, while other common species include whitegrass, marsh fern (*Thelypteris palustris*), beggar’s-ticks (*Bidens frondosa*), false-nettle, wild mint (*Mentha arvensis*), and the shrub winterberry holly.

Locally, quantitative data has been collected from sites described as Lakeside Floodplain Forest at Drowned Lands near the mouth of the Poultney River (Sorenson et al. 1998) and in Ward Marsh along the Lower Poultney River (Field et al. 2001). At the Ward Marsh site, the narrow Lakeside Floodplain Forest may be grading into the floristically similar Silver Maple-

Sensitive Fern Riverine Floodplain Forest. Based on descriptive information in the New York Natural Heritage Program database, the floodplain forests at Bulwagga Bay and the Ausable Delta in New York appear to high-quality examples of lakeside floodplain forests. The natural community descriptions at all of these sites described silver maple as the canopy dominant, with green ash co-dominant at Ward Marsh; the shrub layers were sparse and a number of herbaceous species were noted including sensitive fern, whitegrass, beggar's-ticks, false-nettle and other species.

Restoration of Ecological Processes and Dynamics in Floodplain Forests

Hydrologic conditions clearly exert a strong influence on the composition and structure of floodplain forests. Mitsch and Gosselink (2000) identified the factors that influence hydrology, and they emphasized these should be taken into account when evaluating sites for restoration. Factors include annual and extreme-event flooding, soil permeability, and soil texture. Mitsch and Gosselink also emphasized that chemistry of groundwater, surface flows, flooding streams and rivers, and soils should be investigated to evaluate and monitor changes in water quality.

The importance of flood regime in influencing species composition and physiognomy was observed by Toner and Keddy (1997). They found that the last day of the first flood and the time of the second flood best predicted whether riparian wetlands were dominated by herbaceous or woody species in the temperate zone. Hupp and Osterkamp (1985) found that hydrological processes were more important than sediment size in explaining the variation in species composition on different fluvial landforms of a southeastern bottomland hardwood forest. While riverine floodplains in the Southern Lake Champlain Valley typically do not contain the diversity of fluvial landforms present in southeastern bottomlands, studies such as that conducted by Hupp and Osterkamp point to the importance of hydrology in structuring vegetation in riparian zones and can shed light on vegetation differences among sites.

Annual variation in hydrology can be critical for influencing the establishment of species within a floodplain. Sorenson et al. (1998) considered the establishment of silver maple on beach ridges in Lakeside Floodplain Forests, noting that in most years thousands of silver maple seeds germinate, but most are killed by the next spring flood. They questioned how many dry years with short duration floods it takes for seedlings to survive and recruit into the sapling layer. More research is needed to better understand establishment dynamics for these forests. Studies in other floodplain forests have found that annual variation does play a critical role in influencing species establishment. For example, in an East Texas riverine floodplain forest, Streng et al. (1989) observed variation over time in woody seedling survival. In most years species with larger seeds, such as oaks, had greater survival even though they produced fewer seeds. By emerging later in the summer, they were not vulnerable to spring floods and damping-off mortality; they also were more tolerant of drought, herbivory and competition. Light-seeded species had low survival during most years, because they emerged earlier and were vulnerable to flood conditions and damping off. Streng et al. found, however, that in years with reduced flooding there was a window of opportunity for the survival of light-seeded species.

The influence of the surrounding landscape and the vulnerability to invasion by exotic species are inter-related ecological processes relevant to floodplain forest restoration. Because of the linear shape (high perimeter:area ratio) of riparian zones, the surrounding uplands can exert an especially strong influence (Naiman and Descamps 1997) that can both facilitate and hinder the achievement of restoration goals. The surrounding uplands can serve as a source of propagules for both native and exotic species and can serve as a source of sediment or pollutants. Conditions are ideal in floodplain forests for the invasion of exotic species. Presence of exposed mineral soil, resulting from the annual cycles of erosion and deposition of sediments, and high light availability, due to characteristically open canopies and narrow width of riparian forests, facilitate the establishment of invasive species; moreover, the river corridor provides a means of dispersal of propagules from upstream (Berger 1993, Thompson and Sorensen 2000).

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APPENDICES

Appendix A. Keywords Utilized in the Bibliographic Database

	KEYWORD	Possible Synonyms	See also
1	Adaptive management		
2	Agriculture		
3	Alder	Alnus	
4	Animals	Fish, Birds	
5	Ash	Fraxinus	
6	Basswood	Tilia	
7	Beaver		Animals
8	Beech	Fagus	
9	Birch	Betula	
10	Birds		Animals
11	Bittersweet	Celastrus	
12	Buckthorn	Rhamnus	
13	Cherry	Prunus	
14	Clay		
15	Community-based	Citizen involvement, Sociology	
16	Competition		
17	Costs	Economics	
18	Cottonwood	Populus	
19	Data analysis		
20	Dead wood		
21	Direct seeding		Seed
22	Dogwood	Cornus	
23	Dynamics		
24	Education		Community-based
25	Elm	Ulmus	
26	Evaluation	Measure success	
27	Experimental design	Sampling	
28	Fertilizer		
29	Fish	Animals	Animals
30	Floodplain	Floodplain forest	Riparian, Wetlands
31	Fragmentation	Patch	Spatial analysis
32	Framework	Restoration protocol	
33	Fungi		
34	Genetics		
35	GIS		
36	Goals		
37	Hemlock	Tsuga	
38	Herbaceous	Herbs	
39	Herbivory		
40	Hickory	Carya	

41 Honeysuckle	Lonicera	
42 Hydrology	Flooding	
43 Invasive species	Exotics	
44 Invertebrates	Insects	Animals
45 Landuse	Agriculture	
46 Maple	Acer	
47 Microtopography		
48 Modeling		
49 Monitoring	Baseline, Sampling	
50 Natural regeneration	Seed dispersal, Succession	
51 Northeastern U.S.	New England	
52 Nursery	Propagation	
53 Oak	Quercus	
54 Old growth		
55 Phalaris	Reed canary–grass	Invasive species
56 Pine	Pinus	
57 Planting		
58 Policies	Regulations	
59 Prioritization		
60 Purple loosestrife (3)	Lytrum salicaria	
61 Reference site		
62 Restoration		
63 Riparian	Floodplain	Wetlands, Floodplain
64 Seed	Seed dispersal/ Seedbank/ Seed collection	Direct seeding
65 Seedling establishment	Seedling survival/ Seedling growth	
66 Site history	History	
67 Site preparation		Planting, Techniques
68 Soil		
69 Spatial analysis	Landscape/ Landscape-scale/ Patch	
70 Spruce	Picea	
71 Succession	Natural regeneration, Seed dispersal	
72 Sustainability		
73 Sycamore	Platanus	
74 Techniques	Methods	
75 Thresholds		
76 Tree shelter	Weed control, Protection	
77 Viburnum		
78 Water quality		
79 Weed control		
80 Wetlands		Floodplain, Riparian
81 Willow	Salix	
82 Witch hazel	Hamamelis	

Appendix B. Published Guidelines and Frameworks for Restoration Planning

A. Society for Ecological Restoration (SER 2002)
Plans for restoration projects include, at a minimum the following:

1. A clear rationale as to why restoration is needed;
2. An ecological description of the site designated for restoration;
3. A statement of the goals and objectives of the restoration
4. A designation and description of the reference;
5. An explanation of how the proposed restoration will integrate with the landscape and its flows of organisms and materials;
6. Explicit plans, schedules and budgets for site preparation, installation and post-installation activities, including a strategy for making prompt and mid-course corrections;
7. Well-developed and explicitly stated performance standards, with monitoring protocols by which the project can be evaluated;
8. Strategies for long-term protection and maintenance of the restored ecosystem.
9. Where feasible, at least one untreated control plot should be included at the project site, for purposes of comparison with the restored ecosystem.

B. Key processes essential for successful integration of restoration into land management (Hobbs and Norton 1996)

1. Identifying processes leading to degradation and decline
2. Develop methods to reverse or ameliorate degradation or decline
3. Determining realistic goals for reestablishing species and functional ecosystems, recognizing both the ecological limitations on restoration and the socioeconomic and cultural barriers to its implementation
4. Developing easily observable measures of success
5. Develop practical techniques for implementing these restoration goals at a scale commensurate with the problem
6. Document and communicate these techniques for broader inclusion in landuse planning and management strategies
7. Monitor key system variables, assess progress of restoration relative to the agreed-upon goals, and adjust procedures if necessary.

C. A systems analytic approach to adaptive ecosystem restoration. (Covington et al. 1999)

1. Clearly diagnose the symptoms and causes of the ecosystem health problem. What are the symptoms that suggest the ecological system has been degraded and what are the underlying mechanisms?
2. Determine reference conditions. What was the condition of the ecosystem before degradation?
3. Set measurable ecological goals. How close to reference conditions do you intend to get? How will you know if you are moving in the right direction?
4. What factors are most limiting to the restoration process?
5. Develop alternative ecosystem restoration hypotheses.
6. Design restoration treatments that will allow you to test the alternative hypotheses.

7. Monitor ecosystem conditions and evaluate hypotheses.
8. Feed the results back into the design and implementation of ecological restoration treatments – adapting management based on results and changing goals.

D. Three stage design process which is appropriate for restoration projects (based on NRCS, Allen et al. 2001).

1. data collection and evaluation;
2. developing a list of the general project features any structures needed and a preliminary layout of the site. (A number of alternatives might be included such that they can be evaluated based on ecological and economic grounds);
3. development of final design including assessment of data and drawings used in preliminary design, selection of alternatives and development of final layout and generation of a report detailing the final design plan

E. Steps in Ecological Restoration (summarized from Apfelbaum and Chapman 1997)

1. Gathering information: site inventory, document historical and postsettlement land use (from maps, aerial photos, histories) and collect or assemble information from reference sites.
2. Formulating hypotheses: develop hypotheses about composition, structure and function; review technical literature and visit remnants to refine hypotheses.
3. Setting goals: develop quantitative or qualitative goals for each management unit that assess the potential for restoration and specify desired future condition.
4. Implementation: implement management tasks tied to each goal; coordinate with landowners; address off site problems that need to be considered; obtain permits, assemble volunteers.
5. Monitoring Program: measure pre-restoration conditions as baseline; select parameters that measure progress toward goals; determine methods to measure parameters and the timing, frequency and intensity of measurements.

F. Society of Ecological Restoration Guidelines for conceptual planning (Clewell et al. 2000)

1. Identify project site locations and boundaries
2. Identify ownership
3. Identify need for restoration
4. Identify kind of ecosystem to be restored and type of restoration project
    a. Repair of damaged ecosystem;
    b. creation of a new ecosystem of the same kind to replace one that was entirely removed;
    c. creation of another kind of regional ecosystem to replace one which was removed;
    d. creation of a replacement ecosystem if previous regional ecosystem cannot be supported;
    e. creation of a replacement ecosystem because no reference system exists.
5. Identify restoration goals pertaining to cultural and social values.
6. Identify physical site conditions in need of repair.
7. Identify stressors in need of regulation or re-initiation.
8. Identify biotic interventions that are needed.

9. Identify landscape restrictions, present and future.
10. Identify project funding sources.
11. Identify labor sources and equipment needs.
12. Identify biotic resource needs.
13. Identify the need for securing permits.
14. Identify permit specification, deed restrictions and other legal constraints.
15. Identify project duration
16. Identify strategies for long term protection and management

Preliminary Tasks:

17. Appoint restoration ecologist responsible for technical aspects of restoration
18. Appoint restoration team
19. Prepare a budget to accommodate the completion of preliminary tasks.
20. Document existing project site conditions and describe the biota.
21. Document project site history that led to need for restoration.
22. Conduct pre-project monitoring as needed. (baseline monitoring)
23. Gather baseline ecological information and conceptualize a reference ecosystem from it upon which the restoration will be modeled and evaluated (pre-project documentation may contribute substantially to the reference, guideline #20)
24. Gather pertinent autecological information for key species.
25. Conduct investigations as needed to assess the effectiveness of restoration methods.
26. Decide if ecosystem goals are realistic or if they need modification (guidelines #4 and #5)
27. Prepare a list of objectives to achieve restoration goals.
28. Secure permits
29. Establish liaison with other interested governmental agencies.
30. Establish liaison with public and publicize project
31. Arrange for public participation in project planning and implementation.
32. Install roads and other infrastructure needed to facilitate project implementation.
33. Engage and train personnel who will supervise and conduct project installations tasks.
34. Describe the interventions that will be implemented to attain each objective. (guideline 27; detailed instructions)
35. State how much the restoration can be accomplished passively (if passive restoration is not realized, then additional interventions must be prescribed, guideline #47).
36. Prepare performance standards and monitoring protocols to measure the attainment of each objective.
37. Schedule the tasks needed to fulfill each objective
38. Procure equipment, supplies, and biotic resources.
39. Prepare a budget for installation tasks, maintenance events and contingencies.

Installation Tasks

40. Mark the boundaries and secure the project area.
41. Install monitoring features.
42. Implement restoration objectives

Post-Implementation Tasks

43. Protect site against vandals and herbivore.
44. Perform post-implementation maintenance. (guideline 34)
45. Reconnoiter the project site regularly to identify needs for mid-course corrections.
46. Perform monitoring as required to document the attainment of performance standards (Monitoring if data will be meaningful for decision-making; guideline #45 negates need for frequent monitoring)
47. Implement adaptive management procedures as needed. (plan must have built-in flexibility; new goals (guidelines and objectives #5 and #27 may have to be adopted).

Evaluation

48. Assess monitoring data to determine if performance standards are being met. (See guideline #47)
49. Describe aspects of restored ecosystem that are not covered by monitoring data. (Compliment documentation completed, guideline #20).
50. Determine if project goals were met, including those for social and cultural values (Based on documentation #46 and #49 with respect to goals, guidelines #4 and #5).
51. Publish an account of restoration project and otherwise publicize it.

<u>G. Recommendations for restoring and creating wetlands</u> (Lewis and Erwin, 1995 in Mitsch and Gosselink, 2000)

1. Wetland restoration and creation proposals must be viewed with great care, particularly when promises are made to restore or recreate a natural system in exchange for a permit.
2. Multidisciplinary expertise in planning and careful project supervision at all project levels is needed.
3. Clear, site-specific measurable goals should be established.
4. A relatively detailed plan concerning all phases of the project should be prepared in advance to help evaluate the probability of success.
5. Site-specific studies should be carried out in the original system prior to wetland alteration if wetlands are being lost in the project.
6. Careful attention to wetland hydrology is needed in the design.
7. Wetlands should, in general, be designed to be self-sustaining systems and persistent features of the landscape.
8. Wetland design should consider relationships of the wetland to the watershed, water sources, other wetlands in the watershed, and adjacent upland and deepwater habitat.
9. Buffers, barriers, and other protective measures are often needed.
10. Restoration should be favored over creation.
11. The capability for monitoring and mid-course corrections is needed.
12. The capability for long-term management is needed for some types of systems.
13. Risks inherent in restoration and creation, and the probability of success for restoring or creating particular wetland types and functions, should be reflected in standards and criteria for projects and project design.
14. Restoration for artificial or already altered systems requires special treatment.

15. Emphasis on ecological restoration of watersheds and landscape ecosystem management requires advanced planning.

H. Questions that help determine objectives and the kind of data required for the design, implementation and evaluation of an effective restoration: (Wissmar and Beschta 1998)

1. What physical and biological factors presently limit riparian populations and communities?
2. What geomorphic and hydrological regimes have been historically modified and presently limit the connectivity of riparian an aquatic ecosystems?
3. What native riparian species have been extirpated or displaced?
4. What exotic plant species have invaded the riparian system?
5. What geomorphic and hydrological regimes provide the most favorable future physical habitat and biological conditions?
6. What are the target species or desired future riparian communities?
7. What are the expected recovery times and successional patterns for the riparian communities?

Appendix C. Example Ecological Assessment Parameters

I. Examples of structural, functional and holistic ecological assessment criteria (NRC 1992)

Structural characteristics

A. Water quality – on and off project site; Dissolved oxygen, dissolved salts, dissolved toxics ad other contaminants, floating or suspended matter, pH, odor, opacity, temperature profiles.
B. Soil condition --erodibility; permeability; organic content; soil stablility; physical composition, including particle size and microfauna
C. Geological condition -- surface and subsurface rock and other strata including aquifers
D. Hydrology – quantity of discharge on annual, seasonal and episodic basis; timing of discharge; surface flow processes, including velocities, turbulence, shear stress, bank/stream storage, and exchange processes; ground water flow and exchange processes; retention times; particle size distribution and quantities of bed load and suspended sediment; and sediment flux (aggradational or degradational tendencies) (Rosgen, 1988)
E. Topography – the relief (elevations and gradients) and configuration of site surface features; and project size and location in the watershed, including position relative to similar or interdependent ecosystems or interdependent ecosystems.
F. Morphology—as indicated by shape and form of ecosystem, including subsurface features. For rivers and streams, it includes channel patterns (braided, meandering, straight); bank width- to-depth ratio; meander geometry (amplitude, length, radius of curvature); cross-sectional depth profiles; and riffle-to-pool ratio.
G. Flora and fauna, including density, diversity, growth rates, longevity, species integrity (presence of full complement of indigenous species found on the site prior to disturbance) productivity, stability, reproductive vigor , size- and age-class distribution, impacts on endangered species, incidence of disease, genetic defects, genetic dilution (by nonnative germ plasm) and evidence of biotic stress.
H. Carrying capacity, food web support, and nutrient availability as determined for specific indicator species. (Nutrient availability and nutrient flux patterns are subsumed under "carrying capacity".

Functional characteristics:

A. Surface and ground water storage, recharge and supply.
B. Floodwater and sediment retention.
C. Transport of organisms, nutrients, and sediments.
D. Humidification of atmosphere (by transpiration and evaporation)
E. Oxygen production.
F. Nutrient cycling
G. Biomass production, food web support, and species maintenance.
H. Provision of shelter for ecosystem users.
I. Detoxification of waste and purification of water.
J. Reduction of erosion and mass wastage.
K. Energy flow.

Holistic characteristics

A. Resilience
B. Persistence
C. Verisimilitude

II. Examples of parameters to measure (Westman 1991)

A. Biota: Composition, absolute and relative abundance, gene frequencies, height, density, biomass, nutrient pools.
B. Physical habitat: topographic features, water quality, water quantity, temperature (air, water soil), soil structure, soil/litter nutrients.
C. Ecological Function:
   Biota: productivity and growth rates, nutrient flux, pollutant flux, migration, fire frequency, fire intensity
   Physical: hydrologic flow, nutrient flux, soil movement,
D. Social values: recreational, historical interest, education and scientific interest, aesthetic, health and safety.

Appendix D. Information that Practitioners Need from Restoration Ecologists

1. Thorough ecological inventories of older projects by bioregion to learn what worked and what did not.
2. Ecosystem descriptions, including fauna and functions.
3. Site prioritization.
4. Soil biota studies that compare restoration and reference sites.
5. Common garden and reciprocal transplant studies, in order to understand degrees of ecotypic differentiation within key species.
6. Methods of accelerating the restoration process so that nuisance species don’t take over (role of mycorrhizae, insects, mammals, birds).
7. Inventory animals, especially those important for functions, such as pollination, soil disturbance by burrowing, cavity creation, seed dispersal.
8. Assess value of natural capital gains from restoration relative to cost of restoration; it’s important for policy-makers to have this information.
9. Develop better (statistically dependable and cost-effective) ways to monitor ecosystem functions to evaluate projects.

from Clewell, A., and J. P. Rieger. 1997. What practitioners need from restoration ecologists. Restoration Ecology **5**:350-354.

Appendix E. Eastern Deciduous Forest Restoration Projects of Interest

The following list is a record of especially pertinent restoration projects that were learned about in reviewing the literature.

Essex County, Ontario. A peninsula east of Lake St. Clair, in the Pt. Pelee Area. Restoration project was initiated in 1992; a native plant nursery was growing 41 species by 1994; 60,000 seedlings had been planted by 1997. Information from Colthurst, et al. (1997).

Pennypack Ecological Restoration, 2955 Edgehill Road, Huntingdon Valley, Pennsylvania 19006; tel: (215) 657-0830; www.libertynet.ogr/pert; pert@libertynet.org. 720 acres – working with researchers from Rutgers University and University of PA to accelerate old-field succession without planting. Extensive information on invasive species control.

Philadelphia, Pennsylvania. Restoration of 1.5 ha of eastern mixed mesophytic forest, initiated 1992. Information from Robertson and Robertson (1995). May be same project as Pennypack.

The Big Woods Project, Minnesota. Conservation and restoration collaborative initiated c. 1992; partners include The Nature Conservancy, Minnesota Department of Natural Resources, and Three Rivers Park District. Extensive experience with planting and direct-seeding native hardwood and shrub species. Contact info: Nancy Falkum, The Nature Conservancy, Southeast Office, 328 Central Avenue, Faribault, MN 55021, (507) 332-0525. Richard Peterson, MN DNR, 1400 Cannon Circle, Faribault, MN 55021, richard.peterson@dnr.state.mn.us ; Charlie Evenson, Three Rivers Park District, 12615 County Road 9, Plymouth, MN 22441-1299, cevenson@hennepinparks.org , (Cornett and Evenson 1999); Monitoring studies apparently initiated by Dr. Kathy Shea, St. Olaf College, 1520 Saint Olaf Ave, Northfield, MN 55057 (Stange and Shea 1998).

Tim Simmons, MA Heritage, Restoration Ecologist; tim.simmons@state.mass.us; Tel: 508-792-7270; I was not able to reach him when I called. I assume he is working on restoration projects on the islands, Cape Cod, and elsewhere in Massachusetts.

NATURAL RESOURCES CONSERVATION SERVICE

CONSERVATION PRACTICE STANDARD

RIPARIAN FOREST BUFFER

(Acre)

CODE 391

DEFINITION

An area of predominantly trees and/or shrubs located adjacent to and up-gradient from watercourses or water bodies.

PURPOSES

- Create shade to lower water temperatures to improve habitat for aquatic organisms.
- Provide a source of detritus and large woody debris for aquatic and terrestrial organisms.
- Create wildlife habitat and establish wildlife corridors.
- Reduce excess amounts of sediment, organic material, nutrients and pesticides in surface runoff and reduce excess nutrients and other chemicals in shallow ground water flow.
- Provide a harvestable crop of timber, fiber, forage, fruit, or other crops consistent with other intended purposes.
- Provide protection against scour erosion within the floodplain.
- Restore natural riparian plant communities.
- Moderate winter temperatures to reduce freezing of aquatic over-wintering habitats.
- To increase carbon storage.

CONDITIONS WHERE PRACTICE APPLIES

On areas adjacent to permanent or intermittent streams, lakes, ponds, wetlands and areas with ground water recharge that are capable of supporting woody vegetation.

CRITERIA

General Criteria Applicable To All Purposes

The location, layout and density of the riparian forest buffer will accomplish the intended purpose and function.

Dominant vegetation will consist of existing, naturally regenerated, or planted trees and shrubs suited to the site and the intended purpose.

All buffers will consist of a Zone 1 that begins at the top of the bank or lake shoreline, and extends a minimum distance of 15 feet, measured horizontally on a line perpendicular to the water body.

Occasional removal of some tree and shrub products such as high value trees is permitted in zone 1 provided the intended purpose is not compromised by the loss of vegetation or harvesting disturbance.

Necessary site preparation and planting shall be done at a time and manner to insure survival and growth of selected species.

Only viable, high-quality and adapted planting stock will be used.

Site preparation shall be sufficient for establishment and growth of selected species and is done in a manner that does not compromise the intended purpose.

Livestock shall be controlled or excluded as necessary to achieve and maintain the intended purpose.

Harmful pests present on the site will be controlled or eliminated as necessary to achieve and maintain the intended purpose.

For optimal carbon storage, select plant species that are adapted to the site to assure strong health and vigor and plant the full stocking rate for the site.

Comply with applicable federal, state and local laws and regulations during the installation, operation (including harvesting activities) and maintenance of this practice.

Additional Criteria To Reduce Excess Amounts of Sediment, Organic Material, Nutrients and Pesticides in Surface Runoff and Reduce Excess Nutrients and Other Chemicals in Shallow Ground Water Flow

An additional strip or area of land, Zone 2, will begin at the edge and up-gradient of Zone 1 and extend a minimum distance of 20 feet, measured horizontally on a line perpendicular to the water body. The minimum combined width of Zones 1 and 2 will be 35 feet and consist of trees and shrubs.

Criteria for Zone 1 shall apply to Zone 2 except that removal of products such as timber, fiber, nuts, fruit and forbs is permitted and encouraged on a periodic and regular basis provided the intended purpose is not compromised by loss of vegetation or harvesting disturbance.

Zone 2 will be expanded in high nutrient, sediment, and animal waste application areas, where the contributing area is not adequately treated or where an additional level of protection is desired.

A Zone 3 shall be added to the riparian buffer when adjacent to cropland or other sparsely vegetated or highly erosive areas to filter sediment, address concentrated flow erosion, and maintain sheet flow. The Filter Strip standard (practice code 393) shall be used to design Zone 3. The minimum width of zone 3 will be 15 feet.

Additional Criteria To Provide Habitat For Aquatic Organisms And Terrestrial Wildlife

Width of Zone 1 and/or Zone 2 will be expanded to meet the minimum requirements of the wildlife or aquatic species and associated communities of concern.

Establish plant communities that address the target wildlife needs and existing resources in the watershed.

CONSIDERATIONS

The severity of bank erosion, concentrated flow erosion or mass soil movement and its influence on existing or potential riparian trees and shrubs should be assessed. Watershed-level or contributing area treatment or bank stability activities may be needed before establishing a riparian forest buffer.

When concentrated flow erosion and sedimentation cannot be controlled vegetatively, consider structural or mechanical treatments.

Favor tree and shrub species that are native, non-invasive, or have multiple values such as those suited for timber, biomass, nuts, fruit, browse, nesting, aesthetics and tolerance to locally used herbicides.

Tree and shrub species, which may be alternate hosts to undesirable pests, should be avoided. Species diversity should be considered to avoid loss of function due to species-specific pests.

Plants that deplete ground water should be used with caution in water-deficit areas.

Allelopathic impacts of plants should be considered.

The location, layout and density of the buffer should complement natural features, and mimic natural riparian forests.

PLANS AND SPECIFICATIONS

Specifications for applying this practice shall be prepared for each site and recorded using approved specification sheets, job sheets, technical notes, and narrative statements in the conservation plan, or other acceptable documentation.

OPERATION AND MAINTENANCE

The following actions shall be carried out to insure that this practice functions as intended throughout its expected life.

The riparian forest buffer will be inspected periodically and protected from adverse impacts such as excessive vehicular and pedestrian traffic, pest infestations, pesticides, livestock or wildlife damage and fire.

Replacement of dead trees or shrubs and control of undesirable vegetative competition will be continued until the buffer is, or will progress to, a fully functional condition.

As applicable, control of concentrated flow erosion and sediment deposition shall be controlled by an adjacent filter strip.

Any use of fertilizers, pesticides and other chemicals to assure buffer function shall not compromise the intended purpose.

REFERENCES

Wetland, Woodland, Wildland – A Guide to the Natural Communities of Vermont. Thompson, Elizabeth and Eric Sorenson. Vermont Department of Fish and Wildlife and The Nature Conservancy. 2000. 456pp.

GENERAL SPECIFICATIONS

Procedures, technical details and other information listed below provides additional guidance for carrying out selected components of the named practice. This material is referenced from the conservation practice standard for the named practice and supplements the requirements and considerations listed therein.

Planted Riparian Buffers

PLANTING DENSITIES

Initial plant-to-plant densities for trees and shrubs will depend on their potential height at 20 years of age. Heights may be estimated based on: 1) performance of the individual species (or comparable species) in nearby areas on similar sites, or 2) from plant field guides or other suitable references.

Plant Types/Heights:	Plants per Acre	Plant-to-Plant Spacing in Feet:
• Shrubs less than 10 feet in height	1200 to 450	6 to 10
• Shrubs and trees from 10 to 25 feet in height	450 to 200	10 to 15
• Trees greater than 25 feet in height	450 to 200	10 to 15

When establishing a planted buffer, a minimum of two (2) rows of trees or (2) rows of shrubs should be established alongside the water body for maximum shade, stabilization and nutrient uptake within the desired buffer width. The remaining area of the designated riparian zone should also be planted or left to meet natural regeneration requirements, plantings can be intermixed with open areas treated for natural regeneration and specific wildlife needs. These openings should not exceed 4,356 square feet (1/10 acre) in area. Open areas should not exceed 25% of the remaining planned riparian zone. Stem density for a completed project should be a minimum of 200 plants per acre and should occur in 3 growing seasons. A minimum of 5 tree and shrub species should be included in a completed project.

Naturally Regenerating or Direct Seeded Riparian Buffers

ESTABLISHMENT DENSITIES

A naturally regenerated riparian buffer is considered initially established when plant densities have reached the planted buffer recommended densities for trees and shrubs. A three (3) year growing season period is a reasonable amount of time in which to determine if natural regeneration would take place and be initially established.

Trees and shrubs are considered established when they have begun to dominate herbaceous plants and undesired shrubs that are competing with it for nutrients, water and sunlight.

All areas immediately adjacent to the watercourse should have trees and or shrubs growing near it.

Open areas within the area designed as a buffer should not exceed 1/10th acre in size and should not exceed more than 25% of the total designated buffer area.

CARE, HANDLING, SIZE AND PLANTING REQUIREMENTS FOR WOODY PLANTING STOCK

Planting stock will be cared and handled as described in Standard and Specification 612.

DIRECT SEEDING OF WOODLAND SPECIES

Buffer areas can be direct seeded as described in National Standard and Specification 652.

PREPARATION OF PLANTING

Planting sites shall be properly prepared based on the soil type and vegetative conditions listed in Woodland Site Preparation, Code 490. For sites to be tilled, leave a 3-foot untreated strip at the edge of the bank or shoreline. Avoid sites that have had recent application of pesticides harmful to woody species to be planted. If pesticides are used, apply only when needed and handle and dispose of properly and within federal, state and local regulations. Follow label directions and heed all precautions listed on the container.

Fabric mulch may be used for weed control and moisture conservation for new plantings on all sites, particularly those with pronounced growing season moisture deficits or invasive, weedy species. Refer to Mulching, 484, for installation procedures.

BUFFER WIDTHS

Even minimum buffer widths provide some benefits to the stream ecosystem. In most instances additional width in excess of basic minimums provide less benefits for specific concern the further the distance from the stream or water body. It is best to base buffer widths on a large array of concerns, including social and economic needs of the landowner as well as other non-water quality related concerns, such as wildlife.

Range of Minimum Width for Meeting Specific Buffer Objective (Palone and Todd)

Concern	Range of Widths
Wildlife	15 to 600 Feet
Water Temperature	5 to 75 Feet
Flood Control	100 to 200 Feet
Streambank Stabilization	15 to 60 Feet
Sediment Control	50 to 200 Feet
Nutrient Removal	50 to 200 Feet

BUFFER WIDTH GUIDE FOR SELECTED WILDLIFE SPECIES

Widths below include the sum of buffer widths on one or both sides of water courses or water bodies and may extend beyond riparian boundaries (in such cases refer to Tree/Shrub Establishment, 612, for design of upland forests).

Species:	Desired Width in feet:
• Bald eagle, cavity nesting ducks, heron rookery, sandhill crane	600
• Common loon, pileated woodpecker	450
• Beaver, dabbling ducks, mink, salmonids	300
• Deer	200
• Lesser scaup, harlequin duck	165
• Frog, salamander	100

TABLE 1 - TREE AND SHRUB SPECIES FOR RIPARIAN AREAS

	HEIGHT AT AGE:		TOTAL	Shade Tolerance (1)	Shade Value (2)	Nutrient Uptake (3)	Inundation Tolerance (4)	Soil Saturation Tolerance (5)	Drought Tolerance (6)	Aesthetics (7)	Native Species (8)	Sediment Deposit Tolerance (9)	Root Depth (10)	Wildlife Value (11)
	10	20	HEIGHT	(L=low, M=med., H=high, Y=yes, N=no)										
Common and Sci Names	(feet)	(feet)	(feet)											
Tree (Conifer)														
White Pine (Pinus strobus)	10	24	100	M	M	M	M	M	M	M	Y	M		M
Hemlock (Tsuga canadensis)	8	20	70	M	H	M	L	L	H	H	Y	L		H
White Spruce (Picea glauca)	8	22	80	M	L	M	M	M	H	M	Y	L		M
Black Spruce (Picea mariana)	8	22	70	M	L	M	H	H	M	M	Y	M		M
Tammarack (Larix laricina.)	10	32	60	L	L	M	M	M	L	H	Y	L		H
Northern White-Cedar (Thuja occ)	6	18	60	M	M	M	M	M	M	H	Y	H		H
Tree (Deciduous)														
Silver Maple (Acer saccharinum)	15	50	60	H	H	M	M	M	M	H	Y	M		L
Black Willow (Salix nigra)	12	30	60	H	M	L	H	H	L	L	Y	H		M
Bass Wood (Tilia americana)	18	26	100	H	M	H	M	M	L	M	Y	M		M
Grey Birch (Betula populifolia)	15	25	30	L	L	L	L	M	M	L	Y	L		L/M
White Birch (Betula papyiferia)	15	34	70	L	L	L	L	L	M	H	Y	L		H
White Ash (Fraxinus americana)	18	36	70	L	L	M	L	L	M	M	Y	L		H
Black Ash (Fraxinus nigra)	16	30	60	L	L	M	H	M	L	L	Y	M		H
Box Elder Maple (Acre negundo)	12	30	50	L	M		H	H		L	Y		H	L
Red Maple (Acer rubrum)	15	40	60	H	M		H	H	M	H	Y			H
Swamp White Oak (Quercus bicolor)		30			M		H	H		L	Y			H
Green Ash (Fraxinus pennsylvanica)		50	60		H		M	H		L	Y			M
American Elm (Ulmus americana)			100		H		H	H	M	H	Y			L
Eastern Cottonwood (Populus delt.)			80		H		H				Y		L	M
SHRUB														
Speckled Alder (Alnus rugosa)	6	12	20	M	M	M	M	H	M	L	Y	H	L	H
Red Ozier Dogwood	6	15	15	L	L	M	M	H	M	H	Y	H	L	H
Alternate-leaf Dogwood (Cornus alternifolia)	6	15	20	M	L	M	M	H	M	M	Y	H		
Pussy Willow (Salix bicolor)	6	12	12	M	L	M	H	H	M	M	Y	H		H
Nannyberry (Virburnum lentago)	5	9	30	M	M	M	M	M	M	M	Y	H	L	M
Witch Hazel (Hammenilis viginian)	6	18	20	M	L	M	M	M	M	H	Y	M		
Streamco willow	6	8	12	M	L	M	M	M	M	M	N	H		
Bankers Willow	6	8	12	M	L	M	M	M	M	M	N	H		
Serviceberry (Amelanchier arborea)			30		L					M	Y			H
Erect Willow (Salix eriocephala)											Y		L	
Sand Bar Willow (Salix exiqua)		25		M	M		H				Y		L	L
American Elderberry (Sambucus c.)		12	15	M	L		H			H	Y			H
Smooth Alder (Alnus serrulata)		10	12	M	L		H				Y			M
Gray Dogwood (Cornus racemosa)		10		M	L		M			H	Y		L	H

Silky Dogwood (Cornus amomum)		10	10	M	L		M				Y		L	H
Buttonbush (Cephalanthus occiden.)		8	15	M	L		H			M	Y		M	M

Attributes: (codes include H = High, M = Medium, L = Low, Y = Yes, N = No, with special notes about individual species denoted by a letter, e.g. “a”)

(1) Shade Tolerance. The plant’s capacity to grow in a shaded condition. H = can grow in the shade of an overstory; M = can grow in partial shade; L = needs full or nearly full sunlight.

(2) Shade Value. The density or fullness of shade provided by an individual plant’s crown in a full leaf-out condition. H = provides full shade; M = a partially open crown that provides patchy or incomplete shade; L = a very open crown that provides little shade.

(3) Nutrient Uptake. The plant’s general capacity to use excess nutrients such as nitrate-nitrogen. H = can use large amounts; M = some excess nutrients used; L = plant is a low-nutrient user.

(4) Inundation Tolerance. General capacity of the plant to withstand standing water, low soil aeration conditions. H = can tolerate10 or more days of inundation; M = can tolerate 2-10day events; L = can tolerate 1-day or less of inundation.

(5) Soil Saturation Tolerance. The plant’s capability to grow in near or saturated soil conditions. H = plant can withstand “wet feet;” M = some tolerance to saturated conditions; L = little or no tolerance of water-saturated soil.

(6) Drought Tolerance. The plant’s capability to grow in droughty or dry soil conditions. H = plant can withstand or has physiology to survive droughty periods; M = some tolerance to drought or dry conditions; L = little or no tolerance of dry soil conditions.

(7) Aesthetics. A very general rating (H, M or L) that indicates some aspect of the plant, e.g., flowers, special foliage characteristic, or plant part color, that enhances the appeal or viewing of the planting.

(8) Native Species. Y indicates the plant is native to the state; N indicates it is introduced.

(9) Sediment Deposition Tolerance. H = plant can withstand repeated, deep deposits of sediment; M = plant can withstand repeated, shallow deposits of sediment; L = plant can withstand little or no sediment deposits.

(10) Root Depth. H = plant roots deeply provides stability to the site; M = plant roots to an intermediate depth or variably; L = a shallow rooting plant that providing less site stability.

(11) Wildlife Value. H = trees and shrubs that are a good source of food and/or cover for wildlife; M = trees and shrubs that provide some food and/or cover; L = trees and shrubs that provide very little food and/or cover.

NATURAL RESOURCES CONSERVATION SERVICE

CONSERVATION PRACTICE STANDARD

RESTORATION AND MANAGEMENT OF DECLINING HABITATS

(acre)

CODE 643

DEFINITION

Restoring and conserving rare or declining native vegetated communities and associated wildlife species.

PURPOSE

- Restore land or aquatic habitats degraded by human activity

- Provide habitat for rare and declining wildlife species by restoring and conserving native plant communities.
- Increase native plant community diversity.
- Management of unique or declining native habitats.

Note: NRCS uses the term "wildlife" to include all animals, terrestrial and aquatic.

CONDITIONS WHERE PRACTICE APPLIES

On any landscape which once supported or currently supports the habitat to be restored or managed.

CRITERIA

General Criteria Applicable to All Purposes

- Methods used will be designed to protect the soil resource from erosion.
- Vegetative manipulations to restore plant and/or animal diversity can be accomplished by prescribed burning or mechanical, biological or chemical methods, or a combination of the four.
- Management measures must be provided to control invasive species and noxious weeds in order to comply with state noxious weed laws.
- To benefit insect food sources for grassland nesting birds, spraying or other control of noxious weeds will be done on a "spot" basis to protect forbs and legumes that benefit native pollinators and other wildlife.
- Management practices and activities are not to disturb cover during the primary nesting period in each state. Exceptions could be granted for periodic burning or mowing when necessary to maintain the health of the plant community. Mowing may be needed during the establishment period to control weeds.
- Rotate periodic planned management or other treatments throughout the restored/managed area.
- Where feasible prescribed burning will be utilized instead of mowing.
- Species will be adapted to soil-site conditions.
- Species will be suitable for the planned purpose.
- Seeding rates will be adequate to accomplish the planned purpose.
- Only certified, high quality, and ecologically adapted native seed and plant material will be used.

- Planting dates, and care in handling and planting of the seed or plant material will ensure that established vegetation will have an acceptable rate of survival.
- Site preparation shall be sufficient for establishment and growth of selected species.
- Timing and use of equipment will be appropriate for the site and soil conditions.

CONSIDERATIONS

Confer with other agencies and organizations to develop guidelines and specifications for conserving declining habitats.

In many cases threatened and endangered species or species of concern will benefit from conservation of declining habitats. Follow-up habitat assessments shall be performed on a regular basis.

Haying and grazing will be planned and managed as necessary to achieve and maintain the intended purpose.

All habitat manipulations will be planned and managed according to soil capabilities and recommendations for management will avoid excessive soil loss.

Plant materials centers and commercial growers should be encouraged to develop plant materials for habitat restorations.

PLANS AND SPECIFICATIONS

Specifications for this practice shall be prepared for each habitat type. Specifications shall be recorded using approved specifications sheets, job sheets, narrative statements in the conservation plan, or other acceptable documentation.

OPERATION AND MAINTENANCE

The following actions shall be carried out to insure that this practice functions as intended throughout its expected life. These actions include normal repetitive activities in the application and use of the practice (operation), and repair and upkeep of the practice (maintenance).

Any use of fertilizers, pesticides and other chemicals shall not compromise the intended purpose of this practice.

NATURAL RESOURCES CONSERVATION SERVICE

CONSERVATION PRACTICE STANDARD

TREE/SHRUB ESTABLISHMENT

(Acre)

CODE 612

DEFINITION

Establishing woody plants by planting seedlings or cuttings, direct seeding, or natural regeneration.

PURPOSE

To establish woody plants for forest products, wildlife habitat, long-term erosion control and improvement of water quality, treat waste, reduction of air pollution, sequestration of carbon, energy conservation, and enhance aesthetics.

CONDITIONS WHERE PRACTICE APPLIES

On any area where woody plants can be grown.

CRITERIA

General Criteria Applicable To All Purposes

Species will be adapted to site conditions and suitable for the planned purpose(s).

Planting or seeding rates will be adequate to accomplish the planned purpose.

Planting dates, and care in handling and planting of the seed, cuttings or seedlings will ensure that planted materials have an acceptable rate of survival.

Only viable, high-quality and adapted planting stock or seed will be used.

Site preparation shall be sufficient for establishment and growth of selected species.

Adequate seed or advanced reproduction needs to be present or provided for when using natural regeneration to establish a stand.

Timing and use of planting equipment will be appropriate for the site and soil conditions.

The acceptability and timing of coppice regeneration shall be based on species, age, and diameter.

The planting will be protected from unacceptable adverse impacts from pests, wildlife, livestock damage, or fire.

Each site will be evaluated to determine if mulching, supplemental water or other cultural treatments will be needed to assure adequate survival and growth.

Comply with applicable federal, state, and local laws and regulations during the installation, operation and maintenance of this practice.

Additional Criteria For Improving Or Restoring Natural Diversity

Species selected will be indigenous to the site and will reflect species composition of the desired stands.

CONSIDERATIONS

When underplanting, trees should be planted sufficiently in advance of overstory removal to ensure full establishment.

Use locally adapted seed, seedlings or cuttings. Priority will be given to plant materials that have been selected and tested in tree/shrub improvement programs. All plant materials should comply with a minimum standard, such as the American Nursery and Landscape Association, Forest Service, or state-approved nursery.

Plans for landscape and beautification plantings should consider foliage color, season and color of flowering, and mature plant height.

Where multiple species are available to accomplish the planned objective, consideration should be given to selecting species which best meet wildlife needs.

Tree/shrub arrangement and spacing should allow for and anticipate the need for future access lanes for purposes of stand management.

Residual chemical carryover should be evaluated prior to planting.

Species considered locally invasive or noxious should not be used.

Species used to treat waste should have fast growth characteristics, extensive root systems, capable of high nutrient uptake, and may produce wood/fiber products in short rotations.

For optimal carbon storage, select plant species that are adapted to the site to assure strong health and vigor and plant the full stocking rate for the site.

PLANS AND SPECIFICATIONS

Specifications for applying this practice shall be prepared for each site and recorded using approved specification sheets, job sheets, technical notes, and narrative statements in the conservation plan, or other acceptable documentation.

Plans and specifications will include the following: adapted tree species for the purposes outlined, spacing, planting methods, cultural practices, maintenance requirements, and variations in methods and species between interplanting, underplanting, and planting in open areas. Separate specifications can be prepared for each of these planting methods.

OPERATION AND MAINTENANCE

The following actions shall be carried out to insure that this practice functions as intended throughout its expected life. These actions include normal repetitive activities in the application and use of the practice (operation), and repair and upkeep of the practice (maintenance).

If needed, competing vegetation will be controlled until the woody plants are established. Noxious weeds will be controlled.

Replanting will be required when survival is inadequate.

Supplemental water will be provided as needed.

The trees and shrubs will be inspected periodically and protected from adverse impacts including insects, diseases or competing vegetation, fire and damage from livestock or wildlife.

Periodic applications of nutrients may be needed to maintain plant vigor.